SURFACE WATER - LAKES

LAKE ECOLOGY AND WATER QUALITY

Background and Limnological Investigations

Diamond Lake has been classified by scientists who study lakes (limnologists) as a highly productive water body due to the availability of nutrients that support the growth of aquatic plants. Periods of high algae abundance in the water (algae blooms) at Diamond Lake have been observed since the 1930s (Hughes 1970). Prior to the 1920s, developments at Diamond Lake consisted primarily of unimproved campgrounds. More extensive development began in the 1920s and included construction of a resort and lakeside residences. The construction of residences continued until the mid 1950s and expansion of the campground facilities continued up to 1972. Visitor use of the area has increased dramatically since the area was first developed. By the mid-1960s, Forest Service officials were concerned that nutrient-rich sewage and other wastes generated by Diamond Lake visitors could contribute to an increase in the growth of aquatic plants (eutrophication). Visitor use projections and possible health and aesthetic concerns led officials of the Forest Service to evaluate the waste collection and treatment needs of the Diamond Lake area and a plan was developed for an improved sanitation system (Burgess 1966). The system was designed to accommodate approximately 15,000 lake-visitors per day, including people using the resort, the south-shore area, the trailer court and various picnic sites and campgrounds (USDA Forest Service 1970). The private residences on the western shore of the lake were not included in the waste collection system. These residences rely primarily on septic systems and simple pit toilets for sewage disposal. As part of the waste water diversion system, sewage waters were diverted to waste-stabilization ponds (lagoons) located outside the lake's watershed. In some cases, septic-tank drainfield systems and simple pit-toilets were replaced by vaults which temporarily store wastes until they can be hauled away. The first use of the new facilities occurred in 1970 and by December 1975 all planned connections to the waste water diversion system were completed (Lauer et al. 1979). The Forest Service continues to operate and maintain these sewage diversion and treatment facilities up to the present time.

In 1971, the Forest Service and US Environmental Protection Agency (EPA) signed a Memorandum of Agreement to systematically study Diamond Lake and assess the effectiveness of nutrient diversion on the condition of the lake (Lauer et al. 1979). From 1971 to 1977 the EPA conducted a research program on Diamond Lake to collect limnological information and identify changes that could be attributed to the nutrient diversion. Following this period of study, EPA concluded that the lake's eutrophication rate had not been affected to any significant degree by sewage diversion, and nutrients from human sources represented a minor portion of the lake's total nutrient load. These researchers reported that nutrient enrichment in Diamond Lake was primarily a natural phenomenon, with the majority of nutrients derived from natural sources (Lauer et al. 1979).

Other investigators (Davis and Larson 1976; Meyerhoff et al. 1978; Salinas and Larson 1995; Eilers et al. 1997; Eilers et al. 2001a) reached a different conclusion implicating human activities as major sources of nutrient enrichment which has accelerated eutrophication. Eilers et al. (2001a) concluded that Diamond Lake has experienced significant deterioration in the 20th century and these changes are associated to some extent to shoreline development but correspond more closely with changes in the introduced tui chub (*Gila bicolor*) population.

The Forest Service has continued to support a monitoring program at Diamond Lake to collect limnological and water-quality information. During the summers of 2001, 2002 and 2003, blooms of the toxin producing blue-green algae (cyanobacteria) *Anabaena flos-aquae* occurred at Diamond Lake. The high abundance and associated public health risks of this planktonic (microscopic and suspended in the water) blue-green algae prompted the Umpqua National Forest in cooperation with Oregon Health Division and Douglas County Health Department to close the lake to water contact activities for periods of time during each of these summer seasons.

LAKE MORPHOMETRY AND SEDIMENTS

AFFECTED ENVIRONMENT

Morphometry

Diamond Lake is a large, relatively shallow natural lake with an elliptical shape. The lake bottom slopes gradually to the deepest portion located slightly north of the center (Figure 1). The maximum depth is 14.78 m (48.5 feet) and the mean depth is 6.85 m (22.5 feet). The majority of the lake area is relatively shallow with greater than 80 percent of the lake area being less than 11 m (36.09 feet) deep (Figure 2). The estimated time to refill the lake if it were emptied (hydraulic residence time) is 1.6 years (Johnson et al. 1985). The surface level of Diamond Lake is artificially elevated during the summer by controlling the flow at the outlet. The elevated level in the summer is approximately 0.61 m (2 feet) higher than levels during other seasons. Table 1 presents a summary of lake morphometry data.

The shallowness of Diamond Lake significantly influences its ecological characteristics. Among the factors affected by the relatively shallow water is the ability of light to penetrate to the bottom of a large portion of the lake contributing to the growth of an extensive macrophyte population. Also in shallow lakes, surface turbulence can mix particles including nutrient rich sediments into the water.

Figure 1. Diamond Lake Bathymetry (Source: JC Headwaters, Inc.).

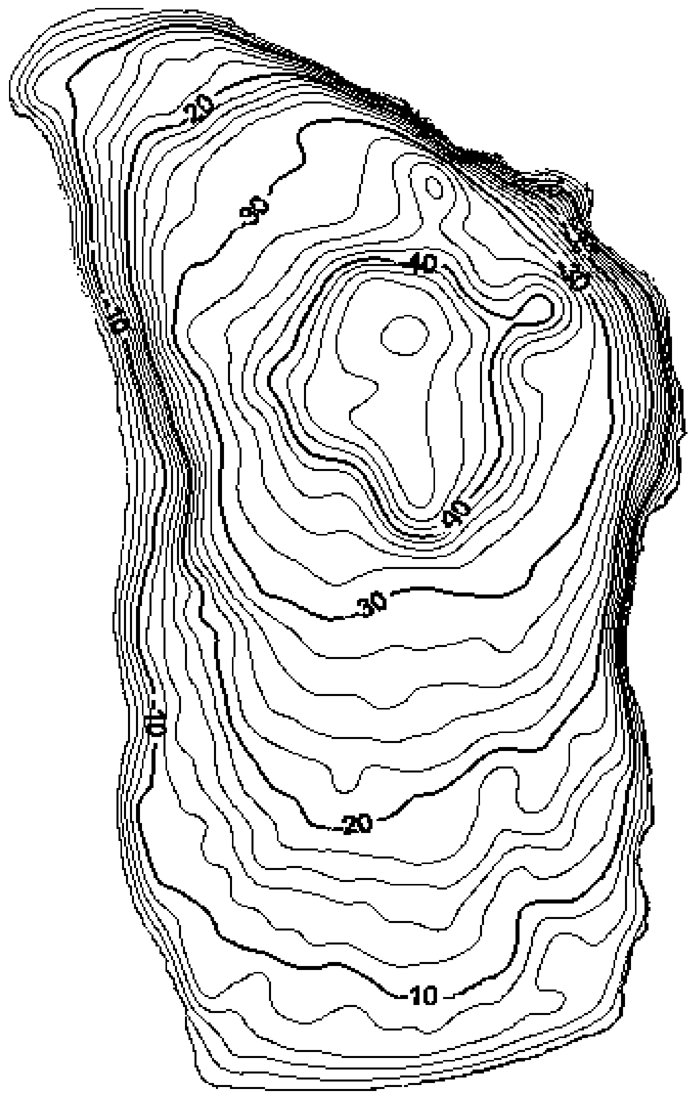

Table 1. Morphometry of Diamond Lake

Attribute	Metric	English
Elevation*	1580 m	5183 ft
Lake Area**	1226 ha	3031 ac
Watershed Area*	136 km^2	55 mi^2
Maximum Depth**	14.78 m	48.5 ft
Mean Depth**	6.85 m	22.5 ft
Volume**	84.00 hm^3	68,099 ac-ft
Precipitation*	140-165 cm	55-65 in

* Source: Johnson et al. 1985
** Source: Eilers and Gubala 2003

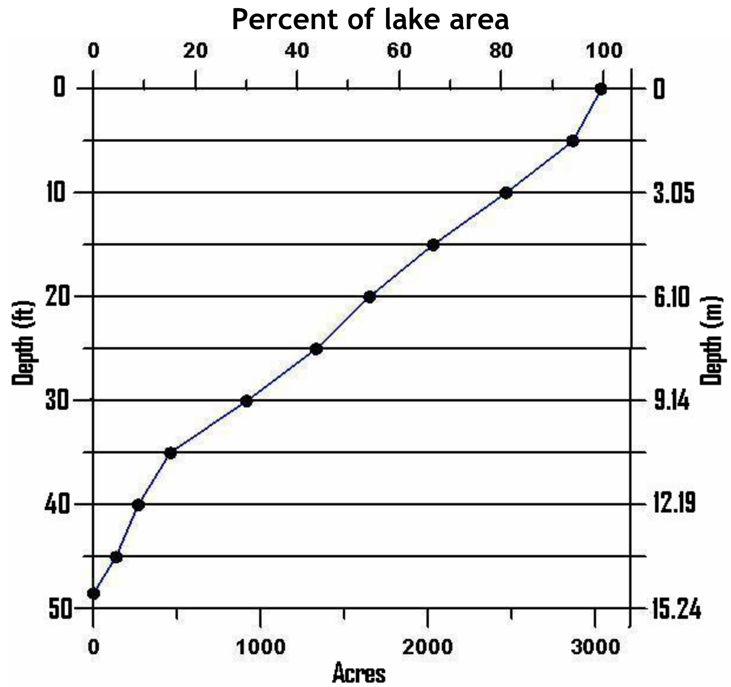

Figure 2. Diamond Lake area by depth (modified from JC Headwaters, Inc. 2003 data).

Sediments

The sediment in the littoral zones of Diamond Lake has been described as flocculent[1], light brown, and often containing rooted aquatic plants (Lauer et al. 1979). The sediments near the center of the lake were described as typically flocculent, gray to brownish organic silt. The organic-rich sediments were observed to provide a habitat highly suitable for macroinvertebrates (Lauer et al. 1979). Eilers et al. (1997) described the sediment profile as relatively uniform, with a high water content, and high organic component. Concentrations of nitrogen, phosphorus, and silicon are high in the sediments.

Davis and Larson (1976) and Meyerhoff et al. (1978) reported an increase in total organic matter in the sediments of Diamond Lake corresponding with an increase in human activities in the watershed. In addition, these investigators observed changes in the planktonic diatom remains in the sediments suggesting an increase in the productivity of the lake associated with an increase in human use. Eilers et al. (2001a) reported an increase in the sediment accumulation rate from approximately 0.01 g/cm/yr around the beginning of the century (~1900) to a rate of approximately 0.03 g/cm/yr from the 1950s through the 1960s and a similar increase again in the late 1980s to mid-1990s (Figure 3). These investigators concluded the sediment accumulation rate has increased approximately four-fold over the long-term baseline values and at least two fold-over values from near the beginning of the century. Although a small portion of the increase in the sediment accumulation rate was

[1] A fine, fluffy mass formed by the aggregation of small insoluble particles that will settle to the lake bottom over time.

associated with watershed development, Eilers et al. (2001a) reported the majority of sediment originated within the lake and corresponds with two time periods of high tui chub abundance. Although the increase in the sediment accumulation rate occurred during a period of warmer than normal air temperatures, these climatic variations likely accounted for only a small part of the observed increase because the rate did not return to background values during cool weather periods.

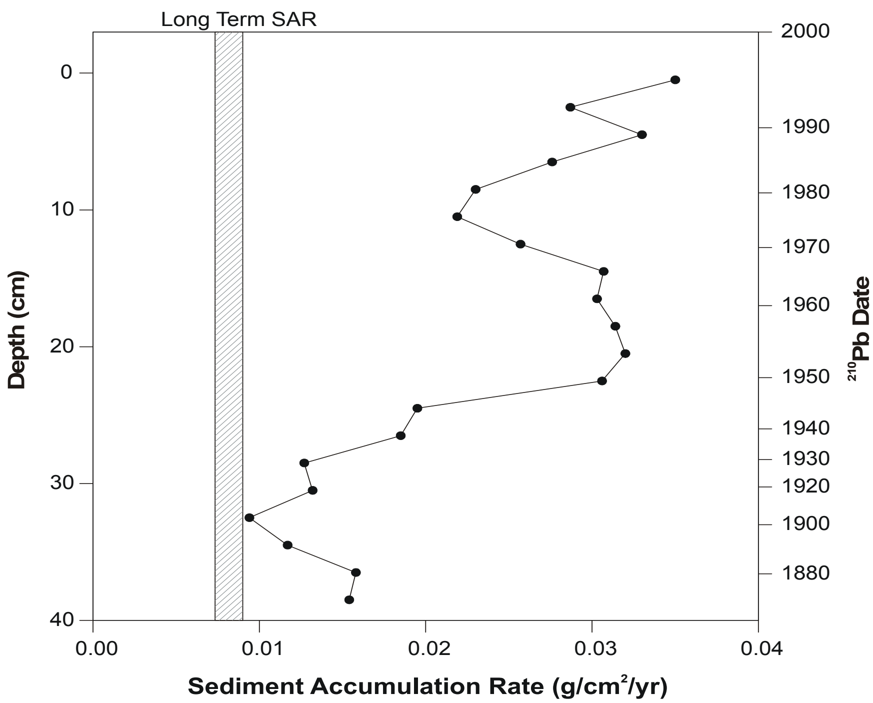

Figure 3. Sediment accumulation rate (Eilers et al. 2001a).

ENVIRONMENTAL EFFECTS ON LAKE MORPHOMETRY AND SEDIMENTS

Direct Effects:
Alternative 1 (No Action) and Alternative 4 would have no direct effect on lake morphometry or sediments since these alternatives do not propose altering the lake level, canal dredging, or wetland expansion. Alternatives 2, 3, and 5 would result in the redistribution of some lake sediments, the greatest effect occurring in the area dredged for canal re-construction and in the wetland expansion area.

Historically and under current operating procedures, the surface of Diamond Lake is artificially elevated during the summer and lowered by 0.61 m (2 feet) during the fall and winter. Under implementation of either Alternative 2, 3, or 5, the surface level of Diamond Lake would be 2.45 m (8 feet) lower than the usual summer level following the fall/winter draw down period estimated to take 4 to 6 months. At the point of maximum draw down during the winter under these alternatives, the lake level would be 1.83 m (6 feet) lower than the usual seasonal water level. The surface area of the lake at the time of maximum draw down would be reduced by 13 percent compared to the normal summer water level and expose approximately 400 acres of sediments along the shoreline. Due to the gently sloping gradient of the southern most portion of the lake bottom, the edge of the water would recede the greatest distance from the normal water edge along this portion of the shoreline. Figure 4 displays the approximate area that would be exposed during the period of maximum draw down. The draw down of the lake surface would be a short-term effect. After the rotenone treatment, the lake would be refilled to the usual seasonal levels.

Alternatives 2, 3, and 5 include excavation of a drainage canal near the current lake outlet. The dredge spoils from the canal construction would be used to expand a wetland along the northwest shore of the lake. Excavation of the drainage canal and in the area of wetland expansion on the northwest shore would be a long-term alteration of a small portion of the lake bottom. High-resolution mapping of the lake bottom by Eilers (2003) revealed that the canal excavated in 1953 to lower the lake level still exists to a significant degree. By following the route of the original canal excavation, the volume of dredge spoils from within the lake would be reduced to a large extent.

Dredging of the outlet channel under Alternatives 2, 3, and 5 could result in reduced water quality in the lake from an increase in turbidity due to disturbance of bottom sediments during the dredging operation. Although these alternatives include the installation of a sediment screen fence around the wetland expansion area, a portion of the fine sediment particles in suspension would be expected to pass through the enclosure and could potentially increase turbidity levels over the entire lake. Due to the small clay content in the sediments and the relatively low degree of exposure to wind generated turbulence, elevated turbidity levels from dredging and wetland expansion activities would be highest in the northwest portion of the lake and would likely subside quickly after completion of the these activities.

Under either Alternative 2, 3, and 5, the operators of the Diamond Lake Resort would request a permit to remove accumulated sediment and trash and repair docks at the resort marina during the low water period following the lake draw down. In addition, the resort operators would conduct similar work to remove old dock structures and moorage material from areas near the South Shore Store. This work would be accomplished using heavy equipment and would affect and area approximately 2/3 of an acre in size. Approximately 750 to 1,000 cubic yards of material would be hauled to an approved waste disposal site. Due to the small area impacted by these activities and no in-water work, no significant direct effects are anticipated from these actions.

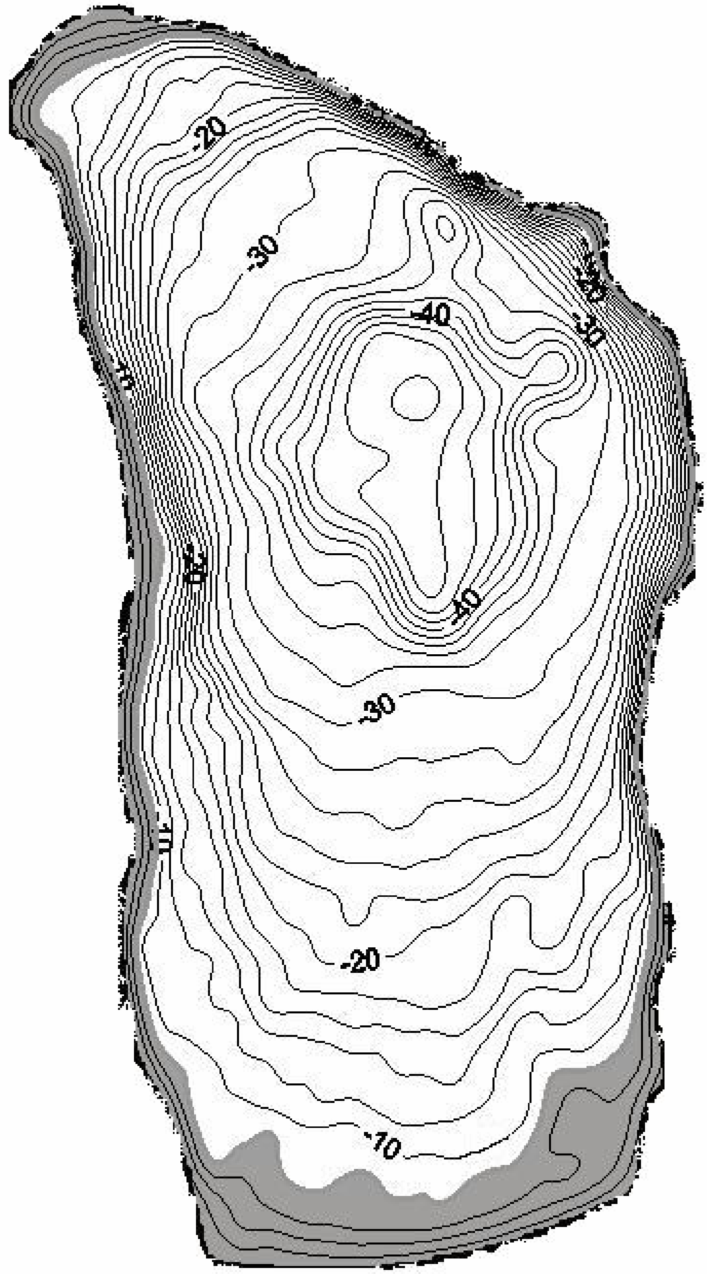

Figure 4. Area of exposed sediment (shown in gray) at time of maximum draw down.

Under Alternatives 2, 3, and 5 the area of wetland expansion would be relatively small compared to the size of the lake and would affect an area estimated to be less than two acres. Turbidity levels in the lake could remain above normal during the draw down operation due to wave action along the shoreline mobilizing and suspending sediments normally under water. However, hydroacoustic data of the lake indicates that the majority of the area exposed to an increase in wave action is composed of relatively hard sediments (Eilers and Gubala 2003). These relatively hard surfaces would be less susceptible to mobilization by wave action reducing the potential and severity of an increase in turbidity.

The sediment accumulation rate would be affected outside of the wetland expansion area as sediments are disturbed by wave action within the draw-down zone and due to settling of suspended sediment originating from activities associated with the dredging and wetland expansion. Following the rotenone treatment under Alternatives 2, 3, and 5 there would be a temporary increase in the rate and quantity of organic material deposited in the sediments as organisms susceptible to the toxic effects of rotenone (primarily fish and zooplankton) die and settle to the lake bottom.

Indirect Effects:
The sediment accumulation rate is also considered to be an indirect effect due to its long-term nature. Since an increase in the sediment accumulation rate has been found to be associated with high tui chub abundance and all alternatives except Alternative 1 would eliminate or severely reduce the tui chub population, the sediment accumulation rate would likely be reduced to some extent by implementation of any of the action alternatives; the reduced sediment accumulation rate would more closely resemble the natural (pre-management) rate and would be viewed as a positive impact. This reduction would likely occur after a period of approximately 3 years under Alternatives 2, 3, and 5. A reduction in the sediment accumulation rate under Alternative 4 would be expected to occur within approximately 7 years from the time mechanical fish removal begins. Alternative 4 may be less effective in reducing the sediment accumulation rate compared to the other action alternatives in the long-term. This is due to both difficulties associated with mechanically maintaining reduced tui chub populations and to new data that creates more uncertainty regarding the potential effectiveness of annual mechanical removal at sustaining water quality and the associated sediment accumulation rate. Although smaller in size than Diamond Lake, Lava Lake (332 acres), is a lake in the Cascade Mountain Range of Oregon that shares a number of features similar to Diamond Lake. Sediment core data collected by Joe Eilers in the fall of 2004 at Lava Lake, where annual mechanical removal of tui chub has occurred since the 1970's, show an increase in the sediment accumulation rate corresponding with the time period of tui chub mechanical removal.

Differences between Alternatives 2, 3, and 5 could occur due to the different fish stocking strategies implemented. Under Alternative 2, the feeding behavior typical of the majority of fish proposed for stocking (rainbow fingerlings) has a higher potential to impact the zooplankton population (particularly large bodied species) and bottom dwelling invertebrates of the lake. High predation of large bodied zooplankton combined with bottom feeding behavior of fish has the potential to affect nutrient distribution and cycling rates and increase overall phytoplankton abundance, possibly leading to an elevated sediment deposition rate. Alternative 3 proposes annual stocking of the lake during the angling season with a type of domesticated rainbow trout. This type of fish would be expected to have a low impact on potential food sources and would not be expected to survive harsh winter conditions. The low feeding rates of these fish would more likely result in conditions favoring high numbers of large zooplankton contributing to an overall reduction in phytoplankton density and a lower sediment deposition rate. Similar to Alternative 2, Alternative 5 includes a stocking strategy for Diamond Lake that includes rainbow fingerlings that would be expected to feed on large bodied zooplankton. Under Alternative 5 however, the number of fingerlings proposed for stocking are lower than Alternative 2 and as a result Alternative 5 would have a lower risk of adverse effects on the zooplankton population. (see sections Aquatic Biology –

Phytoplankton, Zooplankton, and Fisheries for a more complete explanation). Under any action alternative, the effects of fish stocking would be closely monitored using environmental indices to ensure fish populations levels are not contributing to adverse effects on the water quality or other environmental conditions of the lake[2]. If monitoring indicates the potential for adverse conditions to develop, the stocking strategy would be adjusted to maintain or improve environmental conditions.

Under any of the action alternatives a reduction in the sediment accumulation rate would be closer, but not likely equal to the rate before the lake was altered by human activities (positive effect). Under Alternative 1 the sediment accumulation rate would likely remain elevated above rates that existed before the lake was populated with large numbers of tui chub (negative effect).

Cumulative Effects:
Past management activities that were the primary contributors to a cumulative effect on morphometry and sediment accumulation in Diamond Lake were described in the affected environment (i.e. 1950s canal construction, lake-side developments, etc.). Ongoing and reasonable foreseeable actions are not expected to result in adverse effects on these resources.

Under Alternatives 1 and 4, the remains of the previously constructed canal would be filled by sediment deposition over time and no wetland expansion would occur.

Although implementation of Alternatives 2, 3, and 5 include dredging of bottom sediments and wetland expansion, none of the action alternatives propose to expand the dimensions of the canal beyond the size of the original canal constructed in 1953. The sediment deposited in the wetland expansion area would cumulatively add to the natural sediment accumulation in that area and would be an addition to the small quantities of sediment originating from human activities in the watershed. In the majority of the lake, a small increase in the sediment deposition rate could occur during implementation of Alternatives 2, 3, and 5 due to settling of sediment suspended as a result of management activities adding to the existing sediment accumulation rate. As mentioned previously however, very limited quantities of sediments are generated from areas that are impacted by human activities that drain into the lake and most suspended sediment would be likely to be deposited or settle in or near the wetland expansion enclosure. Since all action alternatives would reduce or eliminate the tui chub population and the associated increase in the sediment accumulation rate, any of the action alternatives in combination with other management activities that lower external nutrient loading (e.g. operation of the wastewater diversion system) would cumulatively lower the sediment accumulation rate in the long-term (EIS Tables 9-11 Management Activities in the Cumulative Effects Analysis Area for a complete list of projects). Implementation of contingency plans for all alternatives would also be considered a potential beneficial cumulative effect.

Conclusions:

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[2] An ecologically-based index for guiding fish stocking decisions in Diamond Lake has been developed (Eilers 2003a); components include water chemistry, phytoplankton, zooplankton, and benthos.

In the long-term, all action alternatives would likely result in a reduction in the sediment accumulation rate that would more closely approximate the natural sediment regime. Since Alternatives 2, 3, and 5 would alter only a small area of the lake bottom through canal excavation and wetland expansion, no long-term detrimental effects from this alternation are anticipated. If fish stocking levels were high under Alternatives 2 or 5, the indirect effects could lead to elevated sediment accumulation rate compared to Alternative 3. Among the action alternatives, Alternative 4 would have lower short-term impacts on turbidity and morphometry. However, Alternative 4 may have a reduced probability of achieving and sustaining lowered sediment accumulation rates in the long-term. Preliminary information from Lava Lake that suggests that under some conditions, annual mechanical removal of tui chub can result in a population remaining in an active growth mode resulting in an increase in the sediment accumulation rate (Eilers pers. comm.). As previously stated, the stocking strategy implemented under any of the action alternatives would be based on monitoring of environmental indices to minimize the potential for adverse effects.

Implementation of any of the action alternatives would move toward attainment of Aquatic Conservation Strategy (ACS) objective number 5, "to maintain and restore the sediment regime under which the aquatic ecosystem evolved." Under Alternative 1 (No Action), the sediment accumulation rate would be higher than projections for the action alternatives and would remain elevated above the natural and average historic accumulation rate.

WATER TEMPERATURE AND THERMAL PROPERTIES

AFFECTED ENVIRONMENT

Water temperature is an important factor in the limnological analysis of lakes for several reasons including its influence on the rate of biochemical reactions, and in combination with the intensity of wind and lake geometry, determines the degree surface waters are mixed with deeper water and as a result influences important processes within the lake.

Diamond Lake has sufficient depth that layers of differing water temperature develop on a seasonal basis. As the intensity of solar radiation increases during the spring and early summer months, thermal stratification occurs as a layer of warm surface water called the "epilimnion" develops over deeper, cooler water below. The depth of the epilimnion is directly related to the degree of wind-generated turbulence at the surface. During most summers, the surface waters of Diamond Lake reach a maximum temperature in the early part of August. The water in the deepest portion of the lake remains relatively cool throughout the summer and is referred to as the "hypolimnion". Between these two layers the area of most rapid change in temperature with depth is referred to as the "thermocline" or "metalimnion" (Figure 5).

Freshwater has a maximum density at 4˚C (39.2˚F) and is less dense at temperatures above and below this point (Wetzel 2001). Due to changes in the density of water with temperature, layers of differing temperature become highly resistant to mixing. However, immediately after the melting of winter ice in the spring, temperatures throughout the water column of Diamond Lake become nearly uniform (isothermal). Also in the fall season as air temperatures cool and input of solar radiation decreases, thermal stratification in the lake begins to break down as surface waters cool and temperatures again become isothermal and near the point of maximum density. Under these nearly uniform temperature conditions, resistance to mixing of water at various depths is low and a small amount of wind over the lake can supply sufficient energy to result in mixing of the entire water column of the lake.

During summer thermal stratification, the warmer water in the epilimnion is less dense than the deeper and cooler water of the hypolimnion. Because the layers of differing temperature and density are resistant to mixing, water in the hypolimnion becomes isolated from water near the surface resulting in differences in the chemical and biological properties of these different layers.

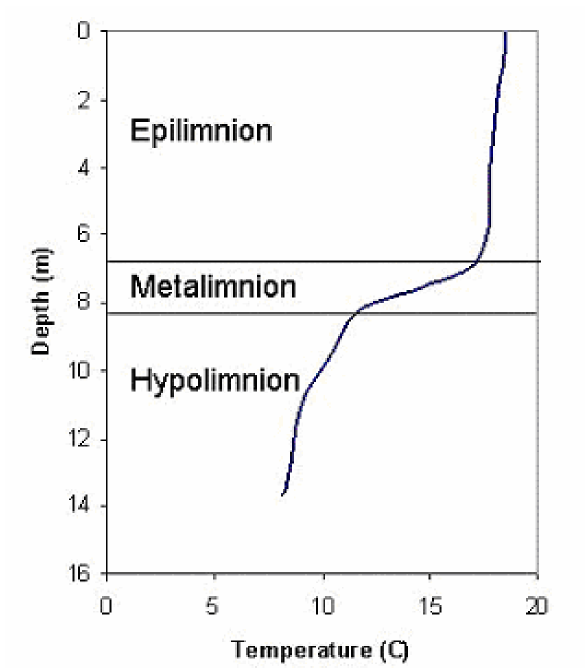

Figure 5. Diamond Lake temperature vertical profile on July 20, 1999 showing epilimnion, metalimnion, and hypolimnion during typical summer thermal stratification.

During most winters, an ice cover forms on Diamond Lake beginning in late December to early January and remains until approximately late March or April. During these periods of ice cover, inverse stratification occurs as surface waters become less dense as they cool below 4˚C (39.2˚F) while deeper water remains near the temperature of maximum density (Figure 6).

Due to its high elevation, development of summer thermal stratification in Diamond Lake generally occurs relatively late in the season (Johnson et al. 1985). Usually, the most prominent stratification occurs during the months of July and August. However, during periods of cool or windy weather, de-stabilization of the stratified layers can occur even during these months (Eilers and Kann 2002). Diamond Lake has been characterized as a dimictic (Lauer et al 1979, Salinas and Larson 1995) meaning that there is complete mixing throughout the water column twice each year, once in the spring, a short time after the melting of winter ice and a second time as the lake cools in the fall. The low stability of the stratification suggests however that the lake behaves more as a system that mixes several times each year (polymictic). One of the consequences of limited duration of stratification is that dissolved oxygen depletion in the isolated water of the hypolimnion generally does not persist for extended periods of time.

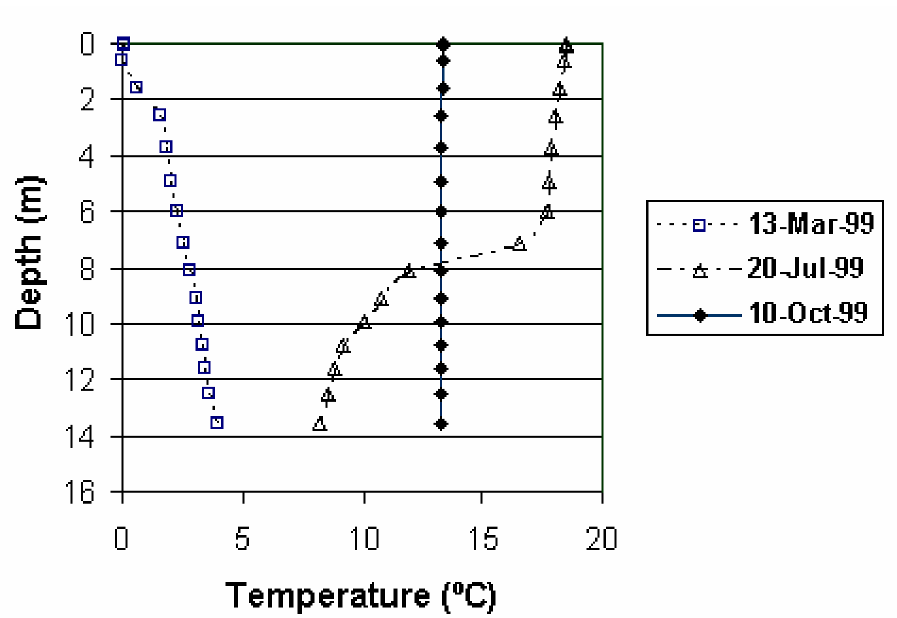

Figure 6. Typical seasonal temperature profiles from Diamond Lake: Winter/spring inverse stratification (March 13, 1999); summer thermal stratification (July 20, 1999), and; fall isothermal profile (October 10, 1999).

ENVIRONMENTAL EFFECTS ON WATER TEMPERATURE AND THERMAL PROPERTIES

Direct Effects:

Under Alternatives 2, 3, and 5 the lake level would be lower by 2.45 m (8 ft) and thus shallower during the summer following the drawdown. Assuming the thickness of the epilimnion would remain the same during this period, wind generated turbulence would likely mix a higher percentage of the lakes total volume during the time when thermal stratification usually inhibits mixing of the deep hypolimnetic water with surface water. As a result, the size and therefore the volume of the hypolimnion would be reduced. In addition, the shallower lake would be more susceptible to mixing of the entire water column resulting in an increase in the frequency of nearly uniform temperatures with depth. Under these conditions the duration of time the lake remains stratified during the summer would be reduced. No direct effects on water temperature or thermal properties would be anticipated under Alternatives 1 or 4.

Indirect and Cumulative Effects:
Since all alternatives would result in maintenance of the historic summer and fall/winter lake levels in the long-term, and no other activities are anticipated that would affect lake water temperature, no indirect or cumulative effects are anticipated under any alternative.

Conclusions:
Although Alternatives 2, 3, and 5 could have short-term effects on the thermal stratification of Diamond Lake during the draw down period, in the long-term all alternatives are not likely to significantly affect the water temperature and thermal properties of the lake and would therefore have a neutral effect on attainment of Aquatic Conservation Strategy Objectives.

WATER CHEMISTRY (DISSOLVED OXYGEN, NUTRIENTS, ALKALINITY and pH)

AFFECTED ENVIRONMENT

The water quality of Diamond Lake varies by season and depth. The majority of variation in water quality is associated with typical changes in seasonal productivity and thermal stratification. Nearly all of the available water quality data for the lake has been collected during the open water period from May through October. Generally the water quality across the surface of the lake is relatively uniform (Salinas and Larson 1995). However, during the summer, variation in some water quality parameters near the surface can occur due to the patchy characteristic of blue-green algae (cyanobacteria) blooms common during this season. Water quality data indicates Diamond Lake is a highly productive water body with sufficient nutrients to support a high phytoplankton biomass during the summer season. Near the surface of the lake during the summer, high phytoplankton primary production results in elevated pH values occasionally greater than 9.3 and can reach values of 9.7 during daylight periods. During the summer seasons of 2001 through 2003, high values of chlorophyll *a* (a measure of algae abundance) and reduced light penetration through the water have been associated with severe blooms of toxin producing cyanobacteria. Despite indications of a decline in water quality in recent years, few water quality parameters indicate clear trends over time. Eilers (2003e) noted that inconsistencies in sampling design and frequency have made it difficult to detect trends in the water quality of Diamond Lake over time. Table 2 displays a summary of water quality data from Diamond Lake collected between 1992 and 2003.

Table 2. Summary of water quality data for Diamond Lake (Source: Salinas and Larson 1992-1994, Salinas 1995-2002, DEQ 2001-2002, Eilers 2003)

	Median	1992-1997	1998-2003
pH*	8.1	7.7 – 9.1	7.4 – 9.7
Chlorophyll a (mg/L)	0.006	0.001 – 0.027	0.005 – 0.064
Total dissolved phosphorus (mg/L)	0.016	0.011 – 0.025	0.008 – 0.030*
Total Phosphorus**(mg/L)	0.020		0.010 – 0.060
Conductivity (µS/cm)	39	33.2 – 51.7	32.1 – 76.6
Nitrate-Nitrogen (mg/L)	0.001	0 – 0.011	0 – 0.516
Ammonia-Nitrogen (mg/L)	0.009	0.001 – 0.527	0 – 0.516

*Derived from maximum measured values
**Data from years 2001-2003

Dissolved Oxygen

Dissolved oxygen (DO) in lake water originates from exchange with the atmosphere and release from plants in the water. DO is required by fish and many other kinds of aquatic life. Although fish species vary in their tolerance to low oxygen, most fish cannot survive at less than 2 mg/L of dissolved oxygen (Wetzel 2001). Diamond Lake is typical of many highly productive lakes in that during periods of summer thermal stratification, high rates of photosynthesis in the epilimnion during the day result in elevated concentrations of oxygen near the surface. However in the deeper water of the lake, dissolved oxygen depletion occurs in both summer and winter.

Dissolved oxygen concentrations in the epilimnion of Diamond Lake are generally near saturation levels[3]. However exceptions to this saturated condition can occur during the summer season. Eilers and Kann (2002) and data from the Oregon Department of Environmental Quality (DEQ) reported dissolved oxygen levels in epilimnetic waters below the expected saturation concentration of approximately 7.5 mg/L during the month of August 2001. Eilers and Kann attributed the lower dissolved oxygen concentrations to the dieback of a dense planktonic cyanobacteria bloom. In the deeper water of the hypolimnion where little light is available for photosynthesis and decaying organic material accumulates, oxygen consuming respiration processes dominate. This results in a depleted oxygen condition during the summer season.

Lauer et al. (1979) reported dissolved oxygen depletion below 10 m depth in both the summer and winter. Values less than 5 mg/L were found below 13 m during the months of July and August of every year between 1971 and 1977 with a low of 0.1 mg/L recorded on August 18, 1977. Lauer et al. observed winter minimum values of 0.5 mg/L and 1.5 mg/L of dissolved oxygen near the lake bottom in February 1972 and March 1975 respectively. Salinas and Larson (1995) reported during July and August in the years 1992 through 1994, concentrations of dissolved oxygen approached 0 mg/L below approximately 12 m. Other studies have reported similar results in recent years indicating declining oxygen concentrations or anoxic[4]

[3] The maximum quantity of dissolved oxygen the water can contain at a given temperature and pressure.
[4] The absence of oxygen

conditions in the hypolimnion during July and August (Eilers and Kann 2002, Salinas unpublished data 2001, 2002, 2003, DEQ 2001 unpublished data). Figure 7 displays seasonal changes in dissolved oxygen concentration with depth during the year 2001 that would be typical of most years.

Oxygen depletion in the hypolimnion results in conditions that favor a series of chemical reactions with the largest effect occurring at the sediment water interface. Important consequences of these reactions are the increased solubility and release of the nutrients phosphorous and silicon from bottom sediments. Increased concentrations of phosphorus in the lake water can contribute significantly to an increase in the growth of algae in the lake. Because the stability of summer thermal stratification in Diamond Lake is relatively low, the duration of stratification is usually insufficient to cause severe oxygen depletion for extended periods of time. However even brief periods of depleted oxygen may be important in contributing to the internal loading of phosphorus from the sediments to the lake water.

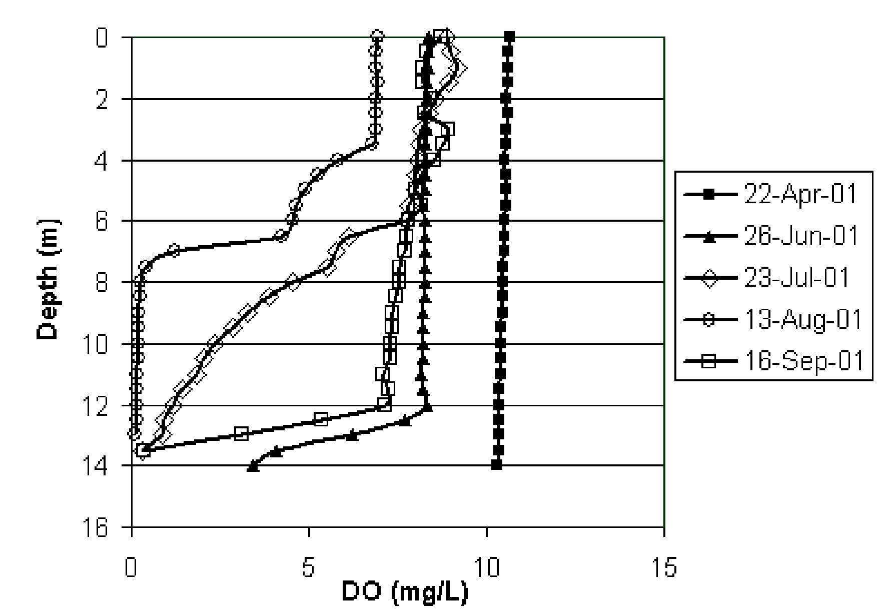

Figure 7. Dissolve oxygen profiles showing seasonal changes in the hypolimnion (data source: Salinas 2001).

Low oxygen concentrations in the deep portion of the lake during periods of strong thermal stratification result in adverse conditions for fish and many other kinds of aquatic life. In addition to the lack of oxygen for oxygen consuming organisms, anoxic conditions in the hypolimnion favors chemical reactions that result in the formation of chemicals toxic to many aquatic organisms, these reactions include the formation of ammonia (NH_3), methane (CH_4), and hydrogen sulfide (H_2S).

Nutrients

Plants require a variety of nutrients to support active growth and photosynthesis. The supply and availability of nutrients in water can limit the growth of aquatic plants (including phytoplankton) in freshwater systems. In addition, the relative abundance of some nutrients can significantly affect the phytoplankton community composition. Nutrient sources for lakes include external loading (groundwater or surface runoff flowing into the lake, direct precipitation, and other sources outside the lake) and from internal loading from sources within the lake.

Although only small concentrations of some nutrients are sufficient to support plant growth, the nutrients phosphorus, nitrogen, and in the case of golden-brown algae (chrysophytes) and diatoms, silica, are required in relatively high quantities. Frequently a correlation is found between the availability of one or more of these nutrients and the productivity of a lake. The form and concentration of these nutrients in the water is affected by both physical and biological processes. The amount of nitrogen and phosphorus in water can change rapidly because they can be taken up by aquatic organisms, stored, transformed, and excreted rapidly and repeatedly (Wetzel and Likens 1991). The availability of these nutrients in Diamond Lake determines to a large degree the productivity of the lake.

Phosphorus

Phosphorus is one of the essential elements required by all living organisms for metabolic processes. Phosphorus is considered to be one of the nutrients that commonly limits algal growth in lakes. Among the variety of phosphorus compounds in lakes, phosphorus can be divided into two general forms, dissolved and particulate. Measurements of total phosphorus include both the phosphorus in solution and in particulate form. The majority of phosphorus exists in a particulate form that is unavailable for uptake by plants. The three main classes of phosphate compounds in aquatic ecosystems include: orthophosphates, condensed phosphates, and organically bound phosphates (MacDonald et al. 1991). Generally, only soluble orthophosphate can be utilized for biological uptake and utilization by phytoplankton and as a result its concentration in lake water provides a good estimation of the availability of phosphorus for algae growth.

The phosphorus budget of lakes includes: (1) external phosphorus loading; (2) internal loading and cycling from within lake processes; and; (3) export downstream or loss to the sediments (Wetzel 2001). Lauer et al. (1979) reported that inflowing streams are the primary external source of phosphorus loading to Diamond Lake. Silent Creek was reported to be the main external source of phosphorus contributing approximately 61 percent of the total measurable load. Short Creek was reported to contribute 22 percent of the external loading during the 1971 to 1977 study period. Intermittent streams were reported to contribute 3 percent, and groundwater sources 10 percent of the external load. Lauer et al. (1979) reported phosphorus in precipitation contributed 4 percent of the total annual phosphorus load. Although Lauer et al. reported low values for groundwater input of phosphorus, current research evaluating the groundwater contribution to the lake may increase this factor in the future as this new data becomes available.

Over the winter months, phosphate generally accumulates in lake water. During the spring, the growth of algae increases the rate of uptake of phosphorus and reduces the concentration of phosphate in the water. As phytoplankton abundance increases in the spring, zooplankton feed on the phytoplankton and also become more abundant and subsequently may be preyed upon by fish. Phosphorus compounds are recycled through the excretion from fish, zooplankton, and bacterial activity into the water where they can again be utilized by phytoplankton.

Loss of phosphorus in the water of Diamond Lake can occur as organisms die and settle to the sediments. Dissolved phosphorus can also be adsorbed onto suspended matter such as fine silt particles and precipitate to the lake sediments. Under oxygenated conditions, phosphorus can also combine with organic matter, metallic ions, and carbonates and precipitate out of the water. Although phosphorus precipitated to the sediments can reduce its concentration in the water, phosphorus can subsequently be recycled back into the water column by in-lake processes. Some phosphorus is exported from Diamond Lake through outflow into Lake Creek, however the overall outflow of phosphorus into Lake Creek is small compared to that retained within the lake. Eilers et al. (2003) estimated that Diamond Lake retains about 50 percent of the total phosphorus inputs into the lake. Based on an analysis of phosphorus concentrations in-inflowing and out-flowing streams, Eilers and Kann (2001) estimated the lake is retaining approximately 0.8 metric tons (~1,764 pounds) of phosphorus from April through September. Figure 8 displays the average total dissolved phosphorus and orthophosphate concentrations in Diamond Lake along with concentrations in the primary inflowing and out-flowing streams. Total dissolved phosphorus exceeds the concentration of orthophosphate in all cases. The nearly complete uptake of available phosphorus by aquatic plants and algae suggests that phosphorus may be a limiting nutrient for periods of time and therefore may limit phytoplankton abundance.

The sediments of Diamond Lake are an important internal source of phosphate.
During summer stratification, the hypolimnion becomes depleted of oxygen. Under these conditions phosphorus is released from the sediments into the hypolimnion (Figure 9). The process of releasing phosphorus from the sediments is referred to as *internal loading*. As previously mentioned, in oxygenated water, phosphorus can combine with oxidized metals, particularly iron, and will precipitate into the sediments, resulting in the removal of phosphorus from the lake water. However, as oxygen levels decline in the hypolimnion, insoluble ferric phosphate molecules in the sediment undergo chemical changes that result in the formation of soluble ferrous phosphate (Wetzel and Likens 1991). The soluble ferrous phosphate is released from the sediments into the hypolimnetic water. When stratification breaks down, the hypolimnetic phosphorus is mixed throughout the entire water column where it is available to support the growth of aquatic plants and algae.

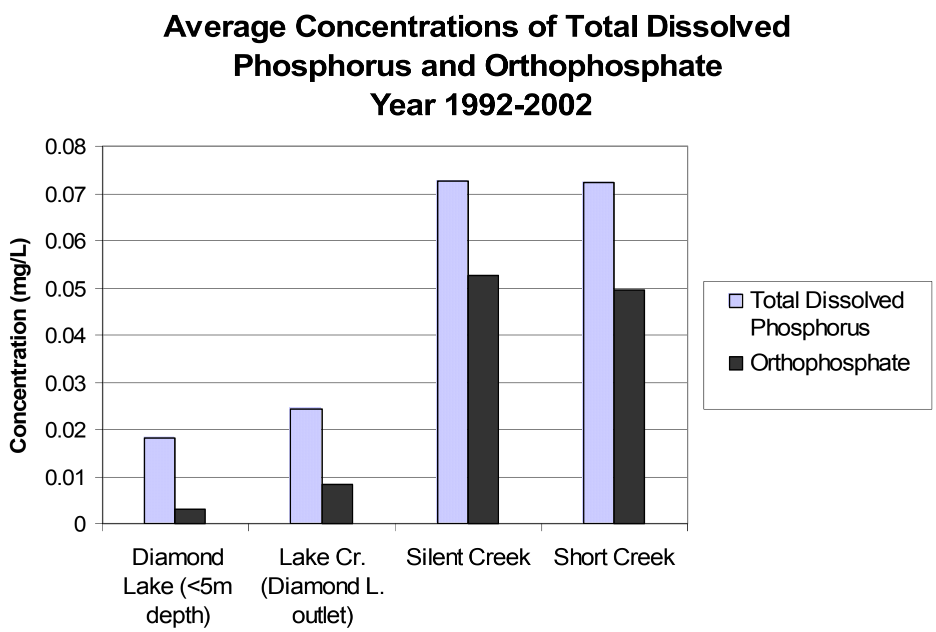

Figure 8. Average concentration of total dissolved phosphorus and orthophosphate for years 1992-2002 (data from Salinas and Larson 1992-1994 and Salinas 1995-2002).

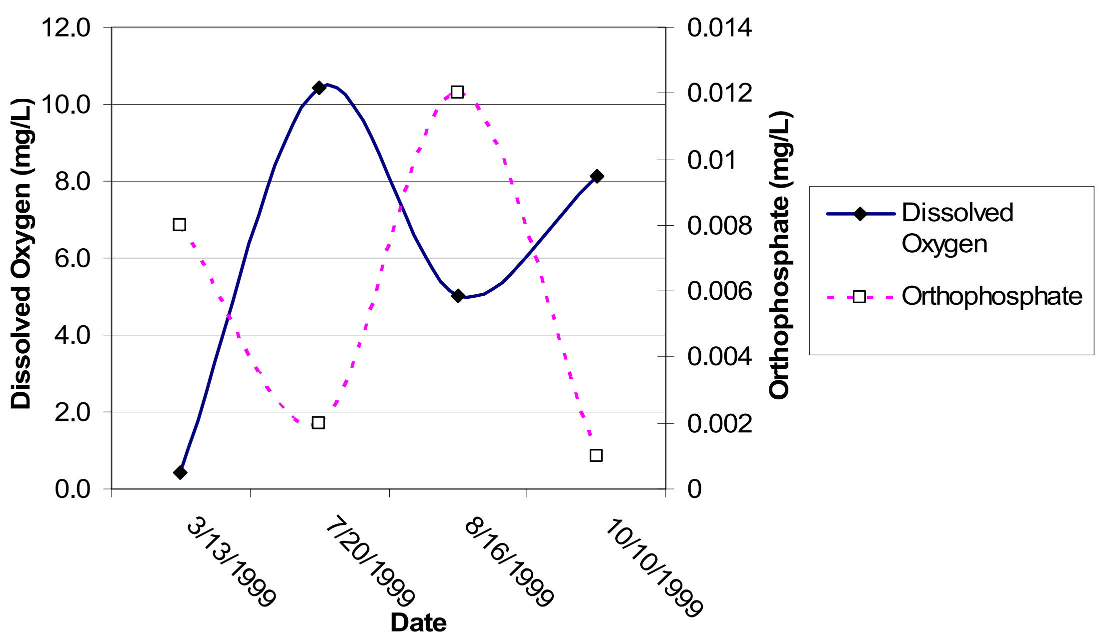

Figure 9. Concentrations of dissolved oxygen and orthophosphate in the hypolimnion of Diamond Lake at 12-14 m depth in year 1999 (data source: Salinas 1999).

Several species of cyanobacteria form spore like resting stages (akinetes) that can over-winter or withstand otherwise harsh environmental conditions in the sediments. During the summer when conditions favor their growth, colonies or cells of these species can migrate from the lake sediments into the water. Pettersson (1998) found that the migration of colonies of the cyanobacterium *Gloeotrichia echinulata* from the sediments to the epilimnion can be a significant pathway for the internal loading of phosphorus. Based on an analysis of akinetes from the sediment of Diamond Lake, Eilers et al. (2001b) concluded that the cyanobacteria population of the lake increased up to 15-fold during the 20th century. Two major increases in the abundance of *Gloeotrichia* were found in the sediment record, evidence of this species first appeared in the sediment record around 1920 following the introduction of trout 10 years earlier, and the second increase occurred around the time of the rotenone treatment in 1954. Eilers et al. (2001b) also reported *Anabaena flos-aquae* and *Anabaena circinalis* were found at low densities prior to the introduction of fish but showed major increases in 1954 and again over the last decade. Deposition of *A. flos-aquae*, *A. circinalis* and *Gloeotrichia* where found to be greatest in the surface sediments suggesting these species could contribute to the internal loading of phosphorus in Diamond Lake.

Fish can also contribute to a redistribution of nutrients including an increase in phosphorus in lake water through bottom feeding behavior followed by excretion of waste products containing nutrients. The high population of tui chub currently found in Diamond Lake are likely to contribute to the redistribution of nutrients through this process. The introduction of additional phosphorus to surface waters can promote the development of algal blooms.

Rooted aquatic macrophytes can play an important role in the availability of phosphorus in lakes. Macrophytes are common in Diamond Lake to a depth of 6 m (Eilers and Gubala 2003). Macrophytes in shallow areas can reduce the amount of phosphorus in the water by stabilizing sediments making them less susceptible to mixing into the water column. Additionally, some species of macrophytes can remove phosphate from lake water by uptake through their foliage and thus compete with algae for nutrients. However the degree to which macrophytes extract nutrients from the water through their foliage or roots varies considerably among species (Wetzel 2001). Concentrations of phosphorus in the pore spaces of lake sediments can be much greater than concentrations of phosphorus in the overlying lake water. Lauer et al. (1979) reported orthophosphate concentrations of 250 µg/L in the pore water of Diamond Lake sediments. Numerous lake studies have shown the majority of macrophytes uptake phosphorus through their roots. (Horne and Goldman 1994). Lauer et al. (1979) suggested uptake of phosphorus by aquatic vascular plants from sediments may be a significant internal source of phosphorus for Diamond Lake. The phosphorus from the sediments incorporated into the plant organic material is eventually released into the water through the death and decay of the plants. Lauer et al. (1979) reported large mats of the macrophytes *Elodea* and *Potomogeton* washed up on the shore of Diamond Lake during storms and concluded that decay of these plants could contribute to nutrients in the water column.

Klotz and Linn (2001) reported that although there are some conflicting reports in the literature regarding the effects of lake draw down on phosphorus release from sediments, studies have shown that drying and freezing of sediments as a result of water level draw down increases the release of phosphorus upon subsequent re-wetting. These investigators attributed this effect to the release of phosphorus from microorganisms that are killed by drying and freezing. Klotz and Linn concluded that site specific characteristics will result in differences between lakes in the relative contribution of sediment drying and freezing to total internal phosphorus loading.

Nitrogen

All living cells require nitrogen in relatively high amounts compared to other nutrients. Nitrogen is an important component of all proteins and is required for most biochemical reactions. It is a major nutrient and its availability frequently is a factor determining the productivity of aquatic systems. Nitrogen is present in aquatic systems in inorganic and organic forms.

The most abundant inorganic form of nitrogen found in streams and lakes is the dissolved gaseous form (N_2). Other inorganic forms generally found in lower concentrations include the combined forms; ammonium (NH_4^+), nitrate (NO_3), and nitrite (NO_2^-). Under some conditions, un-ionized ammonia (NH_3) may also be present. Generally nitrate is the most common form of combined inorganic nitrogen in streams and lakes. Nitrogen in its gaseous form can be utilized for growth only by some cyanobacteria that are capable of nitrogen fixation. This process can be an important source of combined nitrogen in lakes. The preferred form of nitrogen plants utilize for growth is ammonia and as a result under oxygenated conditions is typically rapidly removed from water. Nitrate requires the expenditure of additional energy by plants because it can only be used after it has been transformed by the enzyme nitrate reductase. The waste products of animals that feed on plants or other animals contain ammonia and after these wastes are released into the environment, the ammonia generally is rapidly removed from the water by plants. This recycling of nitrogen in aquatic systems allows for the continued growth of algae in lakes even when other sources of nitrogen are depleted.

Ammonia is highly toxic to fish at relatively low concentrations. However, under oxygenated conditions, ammonia is rapidly removed by aquatic organisms or converted to nitrate. Under acidic to neutral conditions, the formation of ammonium (non-toxic) is favored over formation of ammonia. At high pH values however, the formation of toxic ammonia is favored over ammonium. Because high rates of photosynthesis in productive lakes can result in high pH values, fish die-offs can occur in poorly buffered[5] waters when a critical pH threshold is exceeded leading to the formation of high concentrations of ammonia (Lampert and Sommer 1997). Although Diamond Lake typically has high epilimnetic pH values during the summer, there are no known reports of fish kills associated with high concentrations of ammonia.

Organic nitrogen may be present either as dissolved or suspended particulate matter in water. Organic nitrogen is generally considered to be unavailable for plants until it is converted (mineralized) into nitrate or ammonia. Organic nitrogen is found in a variety of forms including proteins and humic substances. The organic nitrogen concentration is reported as total Kjeldahl nitrogen (TKN) and is determined by the TKN concentration minus the inorganic ammonia concentration. High concentrations of organic nitrogen in water may indicate high rates of production or organic pollution. Human or animal waste, decaying organic matter and live organic material including small algae cells (phytoplankton) can result in high organic nitrogen concentrations in lakes.

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[5] Poorly buffered waters have a chemical makeup with a low ability to neutralize acids or bases without a great change in pH.

The majority of nitrogen present in Diamond Lake originates from three sources: (1) surface and groundwater inflow; (2) precipitation; and (3) fixation of dissolved nitrogen gas (N_2) by cyanobacteria. Lauer et al. (1979) reported that ground water was the primary source of total inorganic nitrogen entering the lake during their study (1972-1977), averaging 80 percent of the external load. In addition, Lauer et al. reported other external sources including Silent Creek and direct precipitation with each of these sources averaging 8 percent. Salinas and Larson (1995) reporting on their study from 1992 to 1994, reported that the inflow to Diamond Lake from Short Creek contained a higher concentration of nitrate than did Silent Creek (mean values of 19.4 and 2.3 µg/L respectively). No studies of Diamond Lake have quantified changes in nitrogen availability due to nitrogen fixation or denitrificaiton[6].

Several studies have evaluated the concentration of inorganic nitrogen in Diamond Lake (Sanville and Powers 1973, Miller et al. 1974, Maloney et al. 1975, Lauer et al. 1979, Salinas and Larson 1995, Eilers 2003b, Eilers et al. 2003). Concentrations of inorganic nitrogen in the lake vary seasonally and with depth. Both nitrate and ammonia are generally found in low concentrations in the epilimnion of Diamond Lake. Reduced concentrations of nitrogen in the near surface water during the summer months has a significant affect on the phytoplankton community composition and result in a shift favoring nitrogen fixing blue-green algae (cyanobacteria). In the deepest portion of the lake during the summer as the dissolved oxygen becomes depleted, ammonia concentrations increase in the hypolimnetic water (Figure 10). As stratification breaks down in the late summer and fall, ammonia from the hypolimnion mixes with surface water where it can stimulate photosynthetic activity.

The external loading of nitrogen into Diamond Lake from streams is primarily inorganic nitrate, whereas in Diamond Lake the majority of nitrogen is found in the organic form (Eilers and Kann 2002) (Figure 11). These differences reflect the high organic nitrogen component associated with the high phytoplankton biomass of the lake. Eilers et al. (2003) estimated that Diamond Lake exports over six times more nitrogen than it receives from the watershed and precipitation. It is assumed that nitrogen fixation by cyanobacteria is the source of the additional nitrogen.

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[6] Conversion of nitrate into nitrogen gases under anoxic conditions.

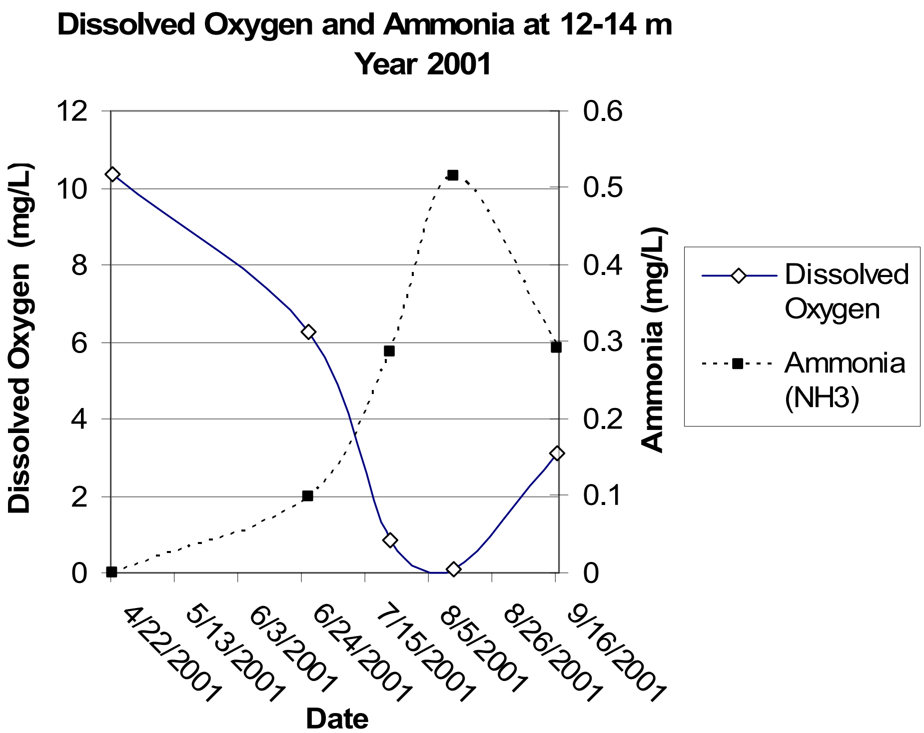

Figure 10. Seasonal changes in the concentration of ammonia (NH_3) and dissolved oxygen at 12-14 m depth in Diamond Lake during the year 2001 (data from Salinas 2002).

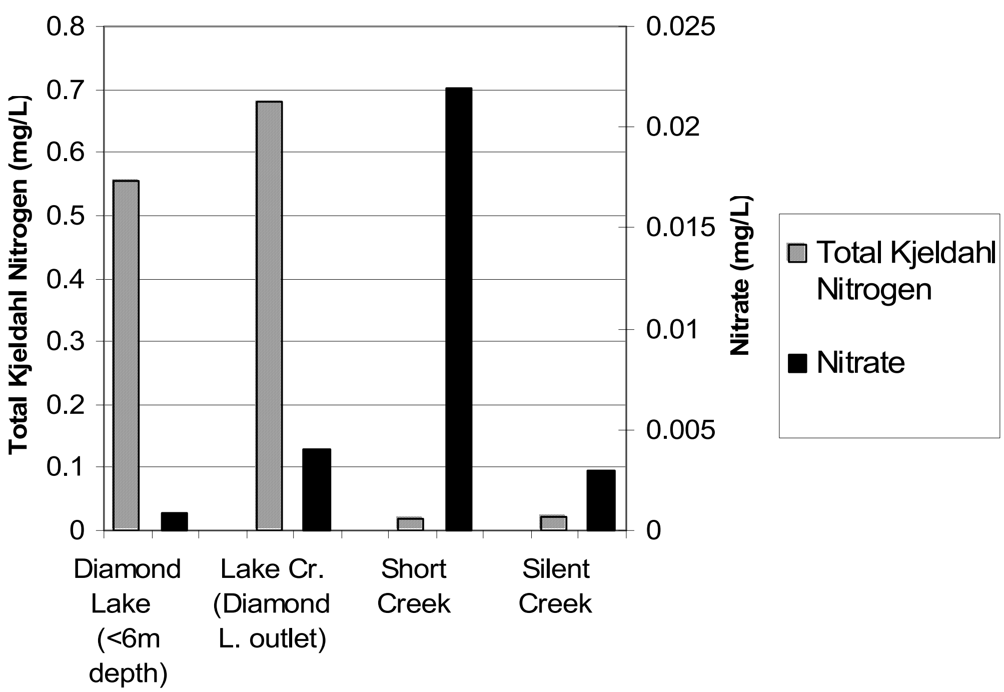

Figure 11. Average values summer season values of total Kjeldahl nitrogen and nitrate in epilimnion of Diamond Lake, and the streams; Lake Creek, Silent Creek and Short Creek (data source Salinas and Larson 1992-1994; Salinas 1995-2002).

Other Nutrients

Plants and animals require relatively small quantities of several nutrients for growth and metabolic processes. Among these are calcium, magnesium, sodium, potassium, sulfur, chloride, iron, silica and very small quantities of several essential metals including manganese, copper, zinc, cobalt and molybdenum (Horne and Goldman 1994). Even though these nutrients are required in relatively small quantities, insufficient concentrations of one or more of these nutrients can limit growth. Diatoms and golden-brown algae (chrysophytes) are an exception to the low requirement of silica (SiO_2). Although silica can become a limiting nutrient in some lakes, concentrations of silica in Diamond Lake always remain high enough to meet the high requirements of diatoms and golden-brown algae. It is possible that some unidentified element(s) could limit algal productivity in Diamond Lake. No known additional studies have been conducted to resolve this issue for Diamond Lake.

Nutrient Loading and Human Activities

Lauer et al. (1979) reporting on the effectiveness of the waste water diversion system and result of studies from 1971 to 1977, found that the diversion system reduced phosphorus loading by 14 percent and nitrogen loading by 18 percent. These investigators found that nutrients from human sources represented a minor portion of the lake's total nutrient load. Lauer et al. (1979) concluded that nutrient enrichment in Diamond Lake is primarily a natural phenomenon, with the majority of nutrients originating from ground water, inflowing stream water and the internal loading of nutrients from sediments.

Eilers (2001a) estimated the nutrient loads from the summer homes near the lake and compared these estimates with other nutrient sources for the lake. He concluded that there is little evidence to suggest that a reduction in nutrients from septic wastes originating from the summer homes near the lake would yield a measurable improvement in water quality. Eilers et al. (2003) estimated that anthropogenic sources account for 16 percent of the total nitrogen load and approximately 1.9 percent of the total phosphorus load. Results of studies of Diamond Lake suggest that the introduction of fish has resulted in significant changes to the lake including a shift in nutrient availability and a decline in water quality (Eilers et al. 2001, Eilers et al. 2003).

Alkalinity and pH

The pH of water is a measure of the activity of hydrogen ions. It is measured on an exponential scale ranging from 0 to 14 and is defined as the negative log of the hydrogen ion concentration. Water with high concentrations of hydrogen ions is acidic with a pH below 7. Water that has low concentrations of hydrogen ions and contains higher concentrations of acid neutralizing chemicals such as hydroxide ions (HO^-), bicarbonate (HCO_3^-), carbonate (CO_3^{2-}), or calcium carbonate ($CaCO_3$) is alkaline and will have a pH greater than 7. Alkalinity is commonly expressed as milligrams per liter of bicarbonate or calcium carbonate.

Near the surface of Diamond Lake during the summer months, high rates of photosynthesis by phytoplankton and macrophytes remove dissolved carbon dioxide and bicarbonate from the water, resulting in an increase in pH. Several studies have determined that during the summer in the epilimnion of Diamond Lake, the pH typically exceeds 8 and during large algae blooms can be greater than 9 (Lauer et al. 1979; Sanville and Powers 1973; Salinas and Larson 1995; Salinas 1996-2002 and ODEQ unpublished data). Epilimnetic pH values typically reach their highest values during July or August. Lauer et al. (1979) measured a maximum pH value of 9.8 near the surface on August 22, 1972. During the summer of 2001, the ODEQ measured a maximum pH value of 9.18 in the surface water of the lake during August and lower values (between 7.5 and 8.5) during May and June (Figure 12). Because Diamond Lake frequently does not meet state water quality standards during the summer due to high eplimnetic pH values, the Oregon Department of Environmental Quality includes Diamond Lake on its list of water quality limited water bodies under section 303(d) of the federal Clean Water Act[7].

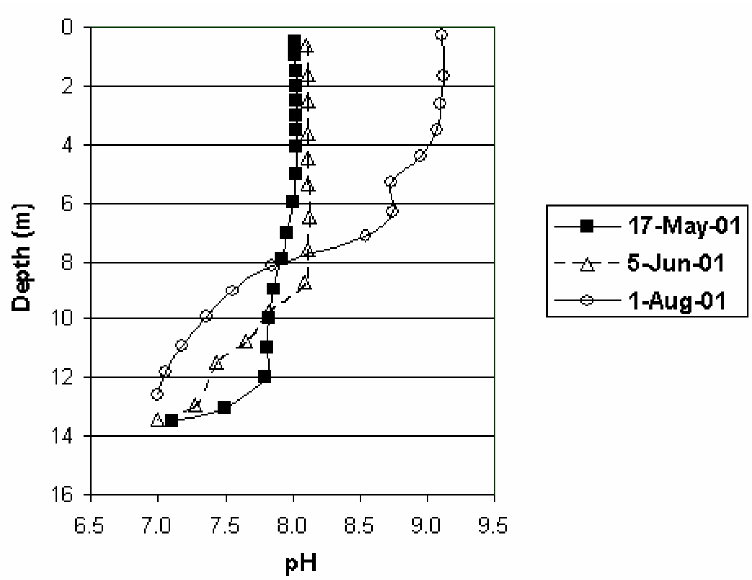

Figure 12. Diamond Lake pH vertical profiles measured by DEQ in 2001.

During stratification in the summer, the deeper hypolimnetic water of Diamond Lake typically has lower pH values due to supersaturation[8] with carbon dioxide generated by decomposition of organic matter. Salinas and Larson (1995) reported slightly acidic conditions (pH 6.5-7.0) near the bottom of the lake during the summers of 1992-1994.

Based on an analysis of the diatom remains in the sediments of Diamond Lake, Eilers et al. (2001a) concluded that the pH of Diamond Lake has increased over pre-development values. Eilers et al. reported that the diatom-inferred pH during the pre-development time period (prior to 1910) averaged 7.9 (±0.31). Although this finding was statistically insignificant (at P = 0.05), an increase in pH during the twentieth century was consistent with increases in the sediment accumulation rate and shifts in the diatom community composition.

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[7] The applicable water body specific pH standard for Cascade Lakes is 6.0 to 8.5 (OAR 340-41-326 (1)(c)).
[8] To become more highly concentrated than is normally possible under given conditions of temperature and atmospheric pressure.

An important consequence of pH in lakes is its effect on the form and concentration of ammonia. Below pH 8, nearly all of the ammonia in freshwater exists in the ionic form ammonium (NH_4^+). This is an important source of nitrogen for aquatic bacteria, algae, and large plants. As the pH increases, the ammonium ion is transformed into un-ionized ammonia (NH_3), a form which can be toxic to aquatic animals including fish. Above pH 10.5, the ammonium ion becomes transformed almost exclusively into the toxic form of un-ionized ammonia. Although high concentrations of ammonia can result in fish kills, none have been reported for Diamond Lake.

Eilers et al. (2003) analyzed the existing conditions of Diamond Lake and used multiple mathematical models to assess current, past, and possible future ecological and water quality conditions in the lake. Modeling by these investigators showed that to meet water quality goals, it would be necessary to remove a high percentage (~90 to 100 percent) of the tui chub from Diamond Lake. However, even under a no-fish scenario for the lake, modeling showed that water quality standards for pH would be exceeded at times but it would be expected that exceedences would be lower than under any scenario with fish present.

Rotenone and Water Chemistry

A review by Bradbury (1986) of several studies that measured the effects of the application of rotenone to lakes concluded that no direct effects on water temperature, dissolved oxygen concentrations, nutrients, carbon dioxide, or pH would be expected post treatment, from the application of rotenone at concentrations used in fisheries management. Bradbury also reported that lakes treated with rotenone generally have shown that algae levels typically increase 4 to 6 fold following a rotenone treatment compared to years without treatment. However, the increase in phytoplankton abundance following rotenone treatment is not due solely to changes in nutrient availability. A significant portion of this increase is due to a large decrease in the grazing pressure of zooplankton (see sections Aquatic Biology – Phytoplankton and Zooplankton).

A variety of factors influence the length of time rotenone remains toxic in the water. The two most important factors determining the rate rotenone degrades are water temperature and sunlight (Bradbury 1986). Rotenone degrades much faster in warm water than cold water (Gilderhus et al. 1986, Dawson et al. 1991). Other factors contributing to the rate of breakdown include the presence of organic debris, turbidity, lake morphology, dilution by inlets and the dosage used. In a wide survey of lakes treated with rotenone, Bradbury (1986) found that the majority of lakes detoxify within 5 weeks from the time of application and lakes in Washington were generally non-toxic to fish after 4 to 5 weeks. In studies conducted by the California Department of Fish and Game (CDFG 1994), the time required for rotenone to degrade to non-toxic levels generally varied from two days to three weeks. Similarly, Finlayson et al. (2000) reported rotenone generally persists from 1 to 8 weeks within a temperature range of 50-68ºF (10-20ºC). Finlayson et al. also reported that following treatment, concentrations of rotenone in the sediments are similar to those in the water however the breakdown of rotenone in the sediments lags 1 to 2 weeks behind that in the water. Rotenolone, a metabolite of rotenone, persists longer than rotenone especially in cold alpine lakes. Finlayson et al. (2000) reported that rotenolone has been detected for as long as six weeks in water <50ºF (10ºC) at elevations >8,000 feet (2,438 m).

The liquid rotenone formulation Noxfish®, contains inert emulsifiers, solvents, and carriers that are important in ensuring the solubility and dispersion of rotenone in water. Waters treated with Noxfish® may contain rotenone, and volatile (xylene, trichloroethylene, toluene, and trimethylbenzene) and semi-volitile (naphthalene, 1-methyl naphthalene, and 2-methyl naphthalene) organic compounds[10]. These volatile and semi-volatile compounds would be expected to persist in the water for less than 3 weeks and in the sediments for less than 8 weeks (Table 3). Many of the inert ingredients of the liquid rotenone formulations (trichloroethylene, naphthalene, and xylene) are present in the fuel of motor boats and as a result are commonly found in lakes where motorized activities occur. Although trichloroethylene is a known carcinogen, Finlayson et al. (2000, p. 189) reported that concentrations in water immediately following treatment with the liquid rotenone formulation would be expected to be below the U.S. Environmental Protection Agency (USEPA) maximum contaminant level[11] in drinking water (0.005 mg trichloroethylene per liter of water) (USEPA 2002b). Finlayson et al. also reported that none of the other materials in the liquid rotenone formulation including xylenes, naphthalene, and methylnapthalenes exceed any water quality criteria or guidelines (based on lifetime exposure) set by the USEPA.

Table 3. Persistence of rotenone and other organic compounds in water and sediment impoundments treated with 2 mg/L rotenone formulation (Finlayson et al. 2000, p. 192).

Compound	Initial water concentration (µg/L)	Water persistence	Initial sediment concentration (µg/L)	Sediment persistence
Rotenone	50	<8 weeks	522	<8 weeks
Trichloroethylene	1.4	<2 weeks	ND*	
Xylene	3.4	<2 weeks	ND	
Trimethylbenzene	0.68	<2 weeks	ND	
Napthalene	140	<2 weeks	146	<8 weeks
1-m-napthalene	150	<3 weeks	150	<4 weeks
2-m-napthalene	340	<3 weeks	310	<4 weeks
Toluene	1.2	<2 weeks	ND	

*ND = Below detection limits

[10] Primarily petroleum-based substances.
[11] Maximum contaminate level is the highest level of a chemical allowed in drinking water. It is an enforceable level under the Safe Drinking Water Act.

ENVIRONMENTAL EFFECTS ON WATER CHEMISTRY

Direct Effects:

Under Alternative 1 (No Action) fish biomass would not be removed from the lake and as a result these nutrients would remain in the lake. Rapid nutrient cycling and nutrient redistribution from the bottom into the water column, associated with a high population of tui chub, would continue to contribute to nutrient availability and contribute to high phytoplankton production. In addition, relatively large summer blooms of blue-green algae (cyanobacteria) would likely continue with the potential for an increase in epilimnetic phosphorus associated with migration of akinetes and colonies from the sediments to surface water. Because Alternative 1 does not include lake draw down, hypolimnetic dissolved oxygen and the associated nutrient release from the sediments would likely remain similar to current conditions.

Alternatives 2, 3, and 5 would result in a lowered lake level during the summer season following the fall/winter draw down. As a consequence, the overall depth of the lake would be reduced and the elevation of the epilimnion would be lower resulting in a corresponding reduction in thickness and volume of the hypolimnion. As is shown in Figure 2, the deeper portions of the lake makeup a relatively small percentage of the total lake bottom. As the lake level is lowered by the draw down, the shallower lake would have a smaller area of bottom sediments exposed to low concentrations of dissolved oxygen during summer thermal stratification. In addition, the shallower lake would increase the probability that wind-generated turbulence could destabilize the thermal stratification and mix the entire water column, potentially reducing the duration of stratification (see discussion under Water Temperature and Thermal Properties). A reduction in the area and duration of depleted oxygen conditions in the hypolimnion of the lake could result in reduced release of phosphorus from the sediments and a reduced probability of high concentrations of ammonia in the hypolimnetic water. These factors could favor a temporary improvement in water conditions (lower pH and improved transparency) during the draw down period. Alternatively, since the volume of the hypolimnion would be lower during the draw down period, it would contain a smaller reservoir of dissolved oxygen and may become anoxic more quickly. This condition would favor the build up of phosphorus and ammonia earlier than may have occurred without the draw down. However, because more frequent mixing would be favored in a shallower lake it is likely that deeper water would be re-oxygenated more frequently. Under any of the alternatives, conditions in the hypolimnion would be determined to a large extent by weather conditions.

During the draw-down period proposed under Alternatives 2, 3, and 5, wave action on exposed sediments along the shoreline would likely disturb fine particles and result in their suspension. In addition, wind or boat-generated turbulence in shallower water would have a higher probability of suspending sediments that would have been deeper underwater if no draw down occurred. These disturbed sediments can release phosphorus, increasing its availability for algal growth. Under oxygenated conditions, however, the extent that these nutrients from sediments would contribute to increased phytoplankton production is likely limited. Under oxygenated conditions, phosphorus is strongly bound to sediment particles.

Although the scouring of lake sediments from wave action or turbulence during the draw down period under Alternatives 2, 3, and 5 would likely be reduced by the stabilizing effect of macrophytes, wave stress on macrophytes could reduce their effectiveness in keeping bottom sediments in place. Additionally the macrophytes within the draw down zone would be exposed to desiccation as a result of the low water level, however the rooted submersed species inhabiting this area would normally be starting to senesce and die back at approximately the time the lake level would be lowered. Therefore, there would be little contribution of additional nutrients to the lake above the typical seasonal amounts from decaying plant material contributing to the nutrient load of the lake under these alternatives.

Under Alternatives 2, 3, and 5, application of rotenone is not expected to directly have significant effects on water temperature, dissolved oxygen concentrations, nutrients, carbon dioxide or pH. There are however numerous potential indirect effects on water chemistry that would result from implementation of these alternatives.

Alternatives 2, 3, and 5 include mechanical fish removal followed by application of rotenone and carcass removal. The removal of fish carcasses from the lake would prevent some of these nutrients from recycling back into the water from the decaying fish. Complete removal of all fish carcasses would not be feasible under either of these alternatives. Results from other lakes treated with rotenone indicate that even when a concerted effort is made to remove fish carcasses, only approximately 20 to 30 percent of the carcasses could be removed (Bradbury 1986). Even if these fish were not exposed to a rotenone treatment, the nutrients contained in their carcasses would eventually be released upon their natural death. In addition, while the fish are alive they continue to contribute phosphorus to the water through excretion. The contribution of the fish carcasses to nutrient loading can be viewed as a matter of timing. Treatment with rotenone would result in an addition of nutrients from the decaying fish as opposed to their contribution to nutrient cycling through feeding and excretion of nutrients.

Under Alternatives 2, 3, and 5, the degree phosphorus released from fish carcasses would contribute to phytoplankton growth depends on the extent that the lake remains oxygenated. Bradbury (1986) concluded that because deep areas of productive lakes typically become oxygen depleted, rotenone-killed fish carcasses generally will not increase the release of phosphorus from the sediments because oxygen levels generally are below 1 mg/L and even without the rotenone treatment, release of nutrients from lake sediments is favored under these conditions. The deepest portion of Diamond Lake typically becomes depleted of oxygen during the summer and winter releasing phosphorus from the sediments. Release of phosphorus from fish carcasses may have only a small effect of increasing nutrient availability in the water due to the existing high rate of nutrient release from the phosphorus-rich sediments of Diamond Lake. The decomposition of the fish carcasses on the lake bottom would create an additional oxygen demand, which could extend the time the sediment-water interface remains depleted of oxygen. A longer duration of depleted oxygen could contribute to elevated phosphorus concentrations in the water. A compensating factor however is that the rotenone treatment would occur during the draw down period when the lake would be approximately 2.45 m (8 ft) shallower than the summer lake level. In addition, the rotenone treatment period (mid-September) coincides with a period of weak stratification. Both of these factors would favor mixing of surface water with the deeper water contributing to the maintenance of oxygenated conditions throughout the water column reducing the potential for phosphorus release from the sediments. Therefore the release of phosphorus from fish carcasses represents a potential impact of limited scale and duration.

The application of rotenone to the lake water under Alternatives 2, 3, and 5 would not be expected to have any direct effects on dissolved oxygen concentrations, nutrients, or pH. A variety of factors influence the length of time rotenone remains toxic in the water. The two most important factors determining the rate rotenone degrades are water temperature and sunlight (Bradbury 1986). Other factors contributing to the rate of breakdown include the presence of organic debris, turbidity, lake morphology, dilution by inlets and the dosage used. In a wide survey of lakes treated with rotenone, Bradbury (1986) found that the majority of lakes detoxify within 5 weeks from the time of application and lakes in Washington were generally non-toxic to fish after 4 to 5 weeks.

Under either Alternative 2, 3, or 5, the operators of the Diamond Lake Resort would request a permit to remove accumulated sediment and trash and repair docks at the resort marina during the low water period following the lake draw down. In addition, the resort operators would conduct similar work to remove old dock structures and moorage material from areas near the South Shore Store. This work would be accomplished using heavy equipment and would affect and area approximately 2/3 of an acre in size. Approximately 750 to 1,000 cubic yards of material would be hauled to an approved waste disposal site. Due to the small area impacted by these activities and no in-water work, no significant direct effects to the water chemistry of the lake are anticipated from these actions.

Alternative 4 would mechanically remove fish from the lake over a 7-year period. Since this would remove fish biomass and the associated nutrients over time, there would be a degree of nutrient loss immediately as fish removal begins and additional losses extending over the entire 7-year fish removal period.

None of the actions in the proposed alternatives would be expected to directly alter pH values.

Indirect Effects:

Under Alternative 1 rapid nutrient cycling would continue to occur due to the high population of tui chub and the associated predation pressure on zooplankton. Under any proposed alternative, the changes in phytoplankton production depend to some extent on the abundance, species, and size composition of the zooplankton population and how these species respond to the fish stocking strategies adopted under different alternatives (see discussion under Aquatic Biology – Zooplankton).

Under the action alternatives, the rate of nutrient cycling in the lake would depend to some extent on the timing and effectiveness of fish removal from the lake. As the fish population declines, there would be a corresponding reduction in the degree nutrients are redistributed from the lake bottom to overlying water. In addition, predation pressure on zooplankton (particularly large bodied species) would be reduced and would contribute to a decline in the rate of nutrient cycling and a lower peak phytoplankton production. These outcomes represent beneficial effects to the lake.

As the lake water level is raised to the normal seasonal level following the draw down under Alternatives 2, 3, and 5, there would likely be an increase in the internal loading of phosphorus due to the re-wetting of previously frozen and dried sediments within the draw down zone. This would increase the availability of phosphorus in the lake water and potentially increase algae production during the spring and summer following return of the lake level to normal seasonal conditions following the draw down period.

The effects of Alternative 4 would occur over a 7 year period as the tui chub population is reduced. The extent to which nutrient cycling changes would be determined by the extent the fish population is reduced and the response of the zooplankton population to reduced predation. As the tui chub population declines over successive years, the peak phytoplankton standing crop and primary production during the summer season would also likely be reduced.

The degree any alternative would affect epilimnetic pH values during the summer season would depend on the degree the alternative affects the growth rate of phytoplankton. Under Alternative 1 (No Action) the nutrient cycling rate would likely remain high during the summer season associated with a high tui chub population. Epilimnetic pH values above 8.5 would be expected to continue during the summer associated with high abundance and growth rates of phytoplankton.

The degree epilimnetic pH values would be reduced during and immediately following mechanical fish removal under all action alternatives would depend on the timing and extent of the fish population reduction. Peak summer season phytoplankton production would likely be lower under Alternatives 2, 3, 4, and 5 if mechanical removal reduces the fish population and associated predation pressure on zooplankton over the spring and summer months resulting in lower pH values during this time. Under Alternatives 2, 3, and 5, following treatment with rotenone, the zooplankton population would be eliminated and since the rotenone treatment is proposed for mid-September, a time when phytoplankton abundance is typically high (particularly diatoms), phytoplankton production and elevated pH values could be greater in the fall than would occur under Alternatives 1 or 4. Studies of other lakes have shown that algae levels typically increase 4 to 6 fold shortly after rotenone treatment compared to levels in years without treatment (Bradbury 1986). However, since the high phytoplankton abundance typically found during the fall would naturally promote high pH levels, any increase in pH compared to non-treatment years would likely be small in scale. As the day length shortens and water temperature decreases as winter approaches, phytoplankton abundance would decline and likely be similar to historic winter levels. In the spring as the phytoplankton population increases rapidly, epilimnetic pH values would rise. The maximum pH values experienced would be a function of nutrient availability, air temperature, the degree of solar radiation, wind, and grazing pressure from the zooplankton. Since the predation pressure on the zooplankton from fish would be temporarily eliminated and the zooplankton would have abundant food resources, the zooplankton would be expected to increase rapidly, particularly large bodied species. This would lead to a reduction in the standing crop and growth of phytoplankton, resulting in lower pH values. The period when lower pH values would be expected to occur under Alternatives 2, 3, and 5 would depend how rapidly the zooplankton population recovers from the effects of the rotenone treatment and the degree of predation on the zooplankton population following fish stocking.

Under Alternatives 2, 3, and 5, peak summer pH values would be expected to decline within 3 years after treatment. The fish stocking strategy adopted under Alternative 3 would result in low predation pressure on zooplankton due to the feeding behavior of the type of salmonid stocked. Therefore Alternative 3 has a somewhat higher potential to result in lower summer pH values than Alternatives 2 and 5 (see sections Aquatic biology – Phytoplankton, Zooplankton, and Fish for additional information). However, under any action alternative, monitoring lake conditions would be required to ensure the fish stocking strategy is maintaining zooplankton-phytoplankton interactions that would promote and maintain acceptable water quality.

Alternative 4 could result in similar results as Alternatives 2, 3, and 5 over an extended period of time. Because the fish population would be reduced over a 7 year period under Alternative 4, changes in the phytoplankton standing crop, primary production, and pH values would likely occur incrementally over this extended time period. Since the tui chub population would be reduced, but not eliminated from the lake under Alternative 4, and since preliminary information from Lava Lake (see FEIS Chapter 3, Environmental Effects on Morphometry and Sediments, Indirect Effects) brings additional uncertainty regarding water quality outcomes (Eilers pers. comm.), there is a higher risk that this alternative would be less effective at sustaining water quality improvements into the future. If mechanical removal resulted in a tui chub population that remains in an active growth mode, implementation of Alternative 4 could result in higher phytoplankton standing crops, primary production and epilimnetic pH values than Alternatives 2, 3, and 5. However it is assumed that the likelihood of achieving or maintaining improvements in the water quality in the long-term would be increased with annual implementation of the described contingency plan.

For Alternatives 2, 3, and 5, if tui chub remain following the rotenone treatment or if/when they are reintroduced into the lake at some unknown point in the future and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, it is assumed that the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.

Under Alternatives 2, 3, and 5 no indirect effects are anticipated as a result of projects proposed by the operators of the Diamond Lake Resort.

Cumulative Effects:

Implementation of any of the action alternatives would be expected to result in a reduced rate of nutrient cycling and a lower standing crop and primary production of phytoplankton associated with the elimination or severe reduction in the tui chub population. The change in nutrient cycling would cumulatively have a positive affect on nutrient availability combined with past projects. Past management activities that were the primary contributors (both positive and negative) to a cumulative effect on water chemistry in Diamond Lake include fish stocking, human developments adjacent to the lake, the 1954 rotenone treatment, and the installation of the waste water diversion system. Ongoing and reasonable foreseeable actions that would contribute to cumulative effects to a small degree include; boat ramp improvements, fish stocking, and water rights. However, as described earlier, the primary negative factor influencing Diamond Lake water quality is believed to be a high population of tui chub, rather than management actions. See EIS Tables 9-11 Management Activities in the Cumulative Effects Analysis Area for a complete list of projects.

Conclusions:

Although there are some short-term impacts as detailed above, in the long-term, implementation of any of the action alternatives would be beneficial toward attainment of Aquatic Conservation Strategy (ACS) Objectives. These alternatives particularly respond to portions of Objectives 4[12] and 5[13] concerning water quality, nutrients, and the natural sediment regime. Action Alternatives 2, 3, and 5 would have the highest probability of providing improved water quality and reduced nutrient loading in the shortest time. The success of Alternatives 2 and 5 at maintaining improved water quality over the long-term would depend on fish stocking levels and to some degree whether monitoring data is effectively utilized to determine appropriate stocking levels. Because the fish stocked under Alternative 3 would be less likely to result in high predation pressure on zooplankton, this alternative may be slightly more favorable at maintaining water quality in the long-term compared to Alternatives 2 and 5. However since the fish stocking strategy proposed under Alternative 5 would have lower numbers of rainbow trout fingerlings that could affect zooplankton populations, Alternative 5 potentially has a lower risk of slowing water quality recovery as compared to Alternative 2. Alternative 4 would also contribute toward attainment of these objectives, however since this alternative would be implemented over a 7-year period, progress toward meeting these objectives would be extended over the 7-year time period and the long-term effectiveness would be less certain.

Table 4 provides a summary of important conclusions and a comparison of the alternatives effect on expected summer pH.

[12] ACS Objective 4 – Maintain and restore water quality necessary to support healthy riparian, aquatic, and wetland ecosystems. Water quality must remain within the range that maintains the biological, physical, and chemical integrity of the system and benefit survival, growth, reproduction, and migration of individuals composing aquatic and riparian communities.
[13] ACS Objective 5 – Maintain and restore the sediment regime under which aquatic ecosystems evolved. Elements of the sediment regime include the timing, volume, rate, and character of sediment input, storage, and transport.

Table 4. Comparison of Alternatives Effects on Summer pH in Diamond Lake

Indicator	Alternative 1 – No Action	Alternative 2 –Rotenone with Put, Grow and Take Fishery	Alternative 3 – Rotenone with Put and Take Fishery	Alternative 4 - Mechanical & Biological	Alternative 5- Modified Rotenone and Fish Stocking
Expected pH	**Short-term** pH expected to remain high due to high phytoplankton primary production	**Short-term** During the first 3 years after treatment, pH potentially would remain high and result in poor water quality	**Short-term** During the first 3 years after treatment, pH potentially would remain high and result in poor water quality	**Short-term** For 7years pH would remain high. Near the end of the 7 years of treatment, pH potentially would decrease and result in improved water quality	**Short-term** During the first 3 years after treatment, pH potentially would remain high and result in poor water quality
	Long-term pH expected to remain high due to high phytoplankton-ton primary production associated with high tui chub population	**Long-term** After 3 years following treatment, pH expected to decrease and result in improved water quality At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.	**Long-term** After 3 years following treatment, pH expected to decrease and result in improved water quality At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.	**Long-term** After 7 years of treatment, pH would potentially be lower for a period of time resulting in improved water quality. However, if annual mechanical removal fails or is stopped, the tui chub population would rebound and subsequent increases in pH and declines in water quality are expected. The likelihood of achieving or maintaining improvements in the water quality in the long-term would be increased with annual implementation of the described contingency plan over time.	**Long-term** After 3 years following treatment, pH expected to decrease and result in improved water quality At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.

LIGHT AND TRANSPARENCY

AFFECTED ENVIRONMENT

Sunlight provides the most importance source of energy to lakes and has profound influences on their ecology. Light is utilized by plants for photosynthesis and its absorption and dissipation as heat affects the thermal properties of lakes. The depth that light is able to penetrate through water is determined by the transparency of the water. Transparency is affected by the amount of suspended organic particles (i.e. plankton), inorganic particles (silt and clay), and dissolved organic matter in the water.

A small portion of the sunlight that reaches the surface of a lake is reflected and the remainder passes through the water where it is refracted[14] or absorbed. The intensity of light decreases exponentially with depth and different wavelengths of light are absorbed or refracted at different rates (Wetzel and Likens 1991). In very clear lakes, the high degree of scattering of blue light and absorption of other wavelengths accounts for the blue appearance. In lakes with an abundance of phytoplankton, green light is largely scattered and as a result the lake will have a green appearance. The transparency of water can be reduced by high concentrations of phytoplankton in highly productive lakes such as Diamond Lake, however, the light scattering effect of phytoplankton is algal cell size dependant. The transparency of water will be greater with the presence of large colonied species like many types of blue-green algae compared to small single cell or small colonied species at the same concentration of chlorophyll a (a measure of phytoplankton abundance).

Diamond Lake does not meet state water quality standards for algae[15] during the summer due to the typically high density of phytoplankton including cyanobacteria. Multiple mathematical modeling of current, past, and possible future ecological conditions in the lake by Eilers et al. (2003) showed that the current biomass of tui chub in Diamond Lake can largely explain the frequency of summer cyanobacterial blooms, although the intensity of the blooms is also largely influenced by weather conditions. The modeling scenarios for Diamond Lake indicated that to meet water quality goals, it would be necessary to remove a high percentage (~90 to 100 percent) of the tui chub from the lake.

An approximation of the transparency of water can be made by using a Secchi disk. This method utilizes a weighted black and white disk, 20 centimeters in diameter. The disk is lowered into the water and transparency is determined by the mean of the depths at which the Secchi disk can no longer be seen when observed from the shaded side of a boat and the depth it reappears while being raised (Wetzel and Likens 1991). Reduced summer Secchi disk measurements in Diamond Lake are primarily due to phytoplankton density. During periods of high phytoplankton abundance, Secchi disk transparency is significantly reduced. Figure 13

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[14] Deflection from a straight path undergone by a light ray in passing at an angle from one median (e.g. air) into another (e.g. water).

[15] The applicable standard states that a three-month (summer) average chlorophyll a value exceeding 0.01 mg/L (for natural lakes) shall be used to identify water bodies where phytoplankton may impair recognized beneficial uses (OAR 340-41-019 (1)(a)(A)).

displays an example of the reduced transparency in Diamond Lake during a period of high phytoplankton abundance during July of 2002.

The changes in both intensity and spectral qualities of light with depth are an important aspect of lake ecology. The depth that light is able to penetrate the water of Diamond Lake is reduced during the summer season due to the dense surface blooms of phytoplankton. Reduced light penetration limits the light availability for macrophytes and deep water dwelling phytoplankton.

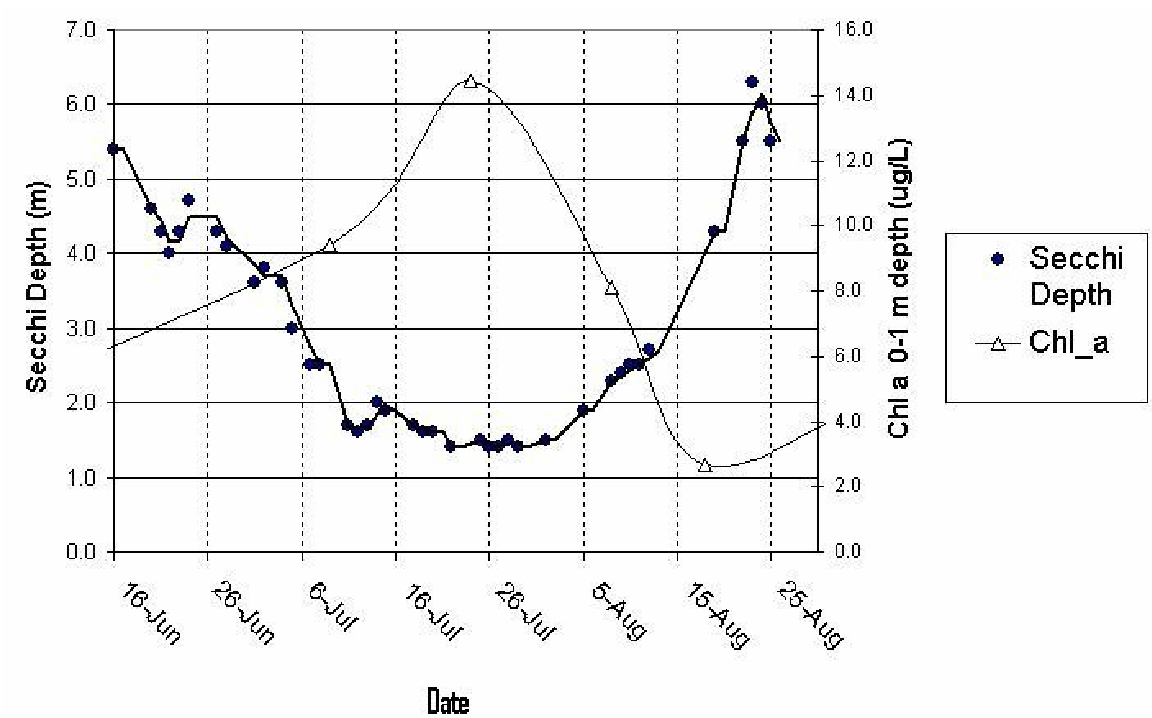

Figure 13. Concentration of chlorophyll *a* (0-1 m depth) and Secchi disk transparency (Summer 2002) (Oregon Department of Fish and Wildlife and Salinas unpublished data).

Plants contain light collecting pigments that are able to use light primarily between the wavelengths of 400 to 700 nanometers (nm). A term used to refer to this range of wavelengths is Photosynthetically Active Radiation (PAR). A light intensity of 1 percent or greater of surface radiation is considered sufficient to support the growth of plants. Monitoring data from Diamond Lake indicate that during periods of high phytoplankton abundance below approximately 6 m in depth, reduced light penetration would severely limit photosynthesis. This finding is consistent with hydroacoustic data indicating that currently macrophytes grow on the bottom of Diamond Lake primarily where the water is less than 6 meters in depth (Eilers and Gubala 2003).

Based on analysis of sediment cores from Diamond Lake, Eilers et al. (2001a) reported that prior to the 20th century, the diatom assemblage of the lake consisted of attached, truly planktonic, and normally attached types in suspension approximately equal in abundance. By the 1930s however, bottom dwelling attached types were declining in number while the proportion of truly planktonic types increased dramatically. These investigators concluded that this change is consistent with the idea that a decrease in transparency was caused by an increase in planktonic diatoms resulting in insufficient light to support the growth of the deeper dwelling attached diatoms. This finding also suggests that area of the lake bottom occupied by macrophytes may have been reduced over the 20th century due to reduced transparency.

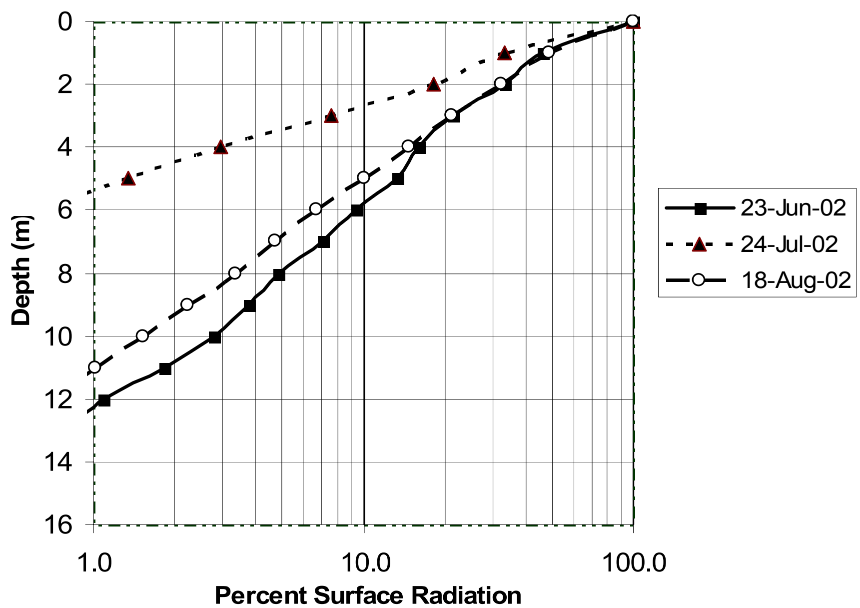

Figure 14. Penetration of photosynthetically active radiation (400-700 nm) in Diamond Lake (Year 2002). Data from Salinas 2002.

ENVIRONMENTAL EFFECTS ON LIGHT AND TRANSPARENCY

Direct Effects:

Under Alternative 1, turbidity levels would not be increased by the disturbance of sediments and high rates of phytoplankton production during the summer months would continue. The high phytoplankton abundance would continue to limit light penetration particularly in mid-summer. Reduced light penetration would limit the ability of macrophytes to occupy the lake bottom below approximately 20 feet (~6m) in depth.

Since Alternatives 2, 3, and 5 would lower the lake level beginning with the fall/winter draw down period, turbidity levels in portions of the lake are likely to be higher due to suspended sediments from canal dredging, wetland expansion activities, and wave disturbance of exposed shoreline sediments. The increase in turbidity would reduce the depth of light penetration and could significantly reduce the potential for photosynthesis in deeper areas of the lake particularly during the summer while the lake is at its low level. Disturbance of lake bottom sediments could result in the release of nutrients that would become available to algae and potentially increase primary production in those portions of the water column where there is sufficient light penetration. Under oxygenated conditions however, the extent that these nutrients from sediments would contribute to increased phytoplankton production is likely to be limited.

Under Alternatives 2, 3, and 5 connected actions proposed by the operators of the Diamond Lake Resort (described in Chapter 2) would not involved any in-water work and would not be expected to have any direct effects on light and transparency in the lake.

Under Alternative 4, no draw down would occur so disturbance of sediments and the associated increase in turbidity associated with that action would not occur. Although extensive netting of tui chub could create minor increases in turbidity, this increase would likely be minimal in intensity and short in duration, quickly subsiding upon completion of the activity.

Indirect Effects:

In the long-term, during the weeks and months following rotenone application, indirect effects would occur under Alternatives 2, 3, and 5. Phytoplankton production would likely rise significantly in Diamond Lake due to a dramatic reduction in grazing pressure from zooplankton and to some degree from a potential increase in nutrient availability from decomposing organisms. As the phytoplankton production increases, light penetration would decrease. The duration of the phytoplankton production increase would be affected by the seasonal changes including reduced temperatures and sunlight. As the fall progresses to winter, the phytoplankton production would decline and transparency would likely be similar to pre-treatment fall conditions. Since the increase in phytoplankton production following the rotenone treatment would coincide with the time of year when the macrophytes senesce and die back, it is unlikely the reduced light availability would have a significant affect on these plants. As spring approaches with longer periods of sunlight and warmer temperatures, under Alternatives 2, 3, and 5, a large bloom of phytoplankton would occur reducing transparency and light penetration to an extent greater than would likely occur under the Alternative 1 or Alternative 4. The standing crop and primary production of phytoplankton during the summer season would remain high until the zooplankton population recovered to the extent that an increase in grazing pressure would lower the phytoplankton abundance. Under the fishless condition that would be present during the spring following treatment under Alternatives 2, 3, and 5, the zooplankton population would likely increase rapidly, creating higher grazing pressure on the phytoplankton. Lowered production and a reduced standing crop of phytoplankton would potentially increase water clarity and promote deeper penetration of light through the water column of the lake. The increase in light penetration during the summer season has the potential to shift a portion of the primary production from the phytoplankton community to attached algae and macrophytes and extend the lake bottom area they are able to occupy. Under Alternatives 2, 3, and 5 recovery of the zooplankton population and the associated effects on phytoplankton would likely occur over a 3-year period. Therefore, a long-term beneficial effect to lake transparency is likely as the peak phytoplankton density is lowered. It is also acknowledged that for these three alternatives, if tui chub remain following the rotenone treatment or if/when they are reintroduced into the lake at some unknown point in the future and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, it is assumed that the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.

Under Alternative 4, the zooplankton population would not be eliminated by the use of rotenone so a shift in the species and size distribution of the zooplankton population would be expected to occur incrementally over a period of about 7-years. Phytoplankton primary production would likely decline over the 7-year implementation period of this alternative in response to changes in grazing pressure and nutrient cycling during the summer period of peak phytoplankton abundance. Since the tui chub population would be reduced, but not eliminated from the lake under Alternative 4, and since the recent preliminary Lava Lakes data (see Chapter 3, Environmental Effects on Water Chemistry, Indirect Effects) brings additional uncertainty regarding long term water quality outcomes (Eilers pers. comm.), there is increased risk that this alternative would be less effective at sustaining improved transparency and other water quality improvements into the future. However, it is assumed that the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.

Under Alternatives 2, 3, and 5 connected actions proposed by the operators of the Diamond Lake resort including removal of accumulated sediment and trash and dock repair are not expected to have any indirect effects.

Under all action alternatives, as phytoplankton production declines, the depth of light penetration would increase, allowing greater opportunities for macrophyte growth moving the lake closer to pre-management conditions.

Cumulative Effects:

Any of the action alternatives have the potential to contribute cumulatively toward the maintenance or improvement in water clarity and reduced phytoplankton primary production. Although the majority of nutrients in Diamond Lake are derived from natural sources, increases in nutrient loading from human activities has the potential to increase the phytoplankton standing crop and primary production resulting in reduced water clarity. Because suspended particles (including phytoplankton) in the water reduce water clarity, activities in the watershed that either reduce external nutrient loading (e.g. the waste water diversion system) or reduce sediment input into the lake (e.g. road surface and roadside stabilization projects) would cumulatively contribute to this goal. See EIS Tables 9-11 Management Activities in the Cumulative Effects Analysis Area for a complete list of past, ongoing, and reasonably foreseeable activities. All current and future projects incorporate Best Management Practices to ensure protection of water quality and beneficial uses through implementation of erosion control techniques and measures to minimize anthropogenic nutrient additions to Diamond Lake. Implementation of contingency plans in the future for all alternatives would be considered a potential beneficial cumulative effect.

Conclusions:

Implementation of any of the action alternatives would be beneficial toward attainment of Aquatic Conservation Strategy (ACS) Objectives. These action alternatives particularly address portions of Objectives 4[16] and 9[17] that address water quality and restoration of habitat to support populations of native plants. Alternatives 2, 3, and 5 would have the highest probability of providing water quality closer to the conditions under which the system evolved in the shortest time (within approximately 3 years). Because the rainbow trout fingerlings proposed for stocking under Alternative 2 have the potential to more aggressively prey on zooplankton (particularly large bodied species), success of this alternative in providing long-term improvement in water quality would depend on utilizing the results of monitoring data to ensure stocking levels do not result in adverse conditions. Although the fish stocking strategy implemented under Alternative 5 would also stock rainbow fingerlings, this Alternative proposes to stock lower numbers and therefore could have a lower risk of leading to adverse changes to water quality an Alternative 2. The type of domesticated trout proposed to be stocked under Alternative 3 have a lower probability of severe predation pressure on zooplankton and bottom dwelling organisms and therefore this alternative has a slightly reduced risk that water quality recovery would be delayed in the long-term compared to other action alternatives. Alternative 4 would also contribute toward attainment of these objectives, however since this alternative would be implemented over a 7-year period, progress toward meeting these objectives would be extended over this time period. Additionally, some evidence suggests that under Alternative 4 there is a risk that the tui chub population may remain in an active growth mode reducing the potential for water quality improvements compared to alternatives 2, 3, and 5. Under all action alternatives, likelihood of long-term success is dependent upon preventing exponential tui chub population growth such as occurred twice in the past. Under any action alternative, a reduction in the peak rate of phytoplankton primary production during the summer would allow light penetration deeper into the lake and would likely promote a macrophyte distribution that more closely resembled the distribution of these species under natural conditions.

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[16] ACS Objective 4 – Maintain and restore water quality necessary to support healthy riparian, aquatic, and wetland ecosystems. Water quality must remain within the range that maintains the biological, physical, and chemical integrity of the system and benefits survival, growth, reproduction, and migration of individuals composing aquatic and riparian communities.

[17] ACS Objective 9 – Maintain and restore habitat to support well-distributed populations of native plant, invertebrate, and vertebrate riparian dependent species.

Aquatic Biology

PHYTOPLANKTON AND PRIMARY PRODUCTION

AFFECTED ENVIRONMENT

PHYTOPLANKTON

The composition and abundance of the phytoplankton[18] population of Diamond Lake varies seasonally. A shift in the phytoplankton population can be influenced by a variety of physical and biological factors. Increased abundance generally occurs during periods of rapid growth (algae blooms) during which different algal types can dominate. There are five phytoplankton groups typically found in Diamond Lake: (1) Bacillariophyta (diatoms); (2) Cyanobacteria (blue-green algae); (3) Cryptophtya (cryptomonads); (4) Chlorophyta (green algae); and, (5) Chrysophyta (golden-brown algae) along with relatively low densities of other types of algae.

The phytoplankton abundance in Diamond Lake is at a minimum during the winter when light and temperatures conditions are low. During the fall and winter, the phytoplankton population is generally dominated by diatoms, cryptophytes, and chrysophytes (Lauer et al. 1979, Aquatic Analysts 1990). As the days lengthen and temperatures warm in the spring, the growth of the phytoplankton population increases and diatom blooms typically develop shortly after the lake ice cover melts. After the initial growth period dominated by diatoms, other types of algae become more common including chlorophytes, chrysophytes and other varieties of diatoms. By mid to late-summer, the phytoplankton species composition shifts to a population dominated by cyanobacteria (Figures 15a and 15b). Historic and recent data from Diamond Lake indicates that during the summer the phytoplankton population is frequently dominated by cyanobacteria from the genus *Anabaena*.

Lauer et al. (1979) recorded high numbers of the diatom *Asterionella formosa* during the spring of each year from 1971 through 1977. The highest concentration of this species, 17,000 cells/mL, was observed on June 4, 1971. Chlorophytes were reported to dominate the phytoplankton on only a few occasions with the highest concentration reported to be 1,200 cells/mL. Lauer et al. (1979) observed that the chrysophyte *Chromulina* sp. appeared to be a much larger part of the phytoplankton community after 1973, although this observation could have been due to a change in counting techniques. This species was the dominant alga in approximately one-half of the samples from 1973 through 1977 and was dominant in all of the 1977 samples, excluding one sample on August 18, 1977 when it was second to *Anabaena*. Salinas and Larson (1995) reported 55 phytoplankton taxa collected during 1992 through 1994. Similar to the results reported by Lauer et al. (1979), Salinas and Larson observed *Chromulina* sp. to be the most dominant taxon based on numerical abundance (measured in cells/mL).

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[18] Plants usually microscopic, comprised primarily of algae, that live suspended in the water.

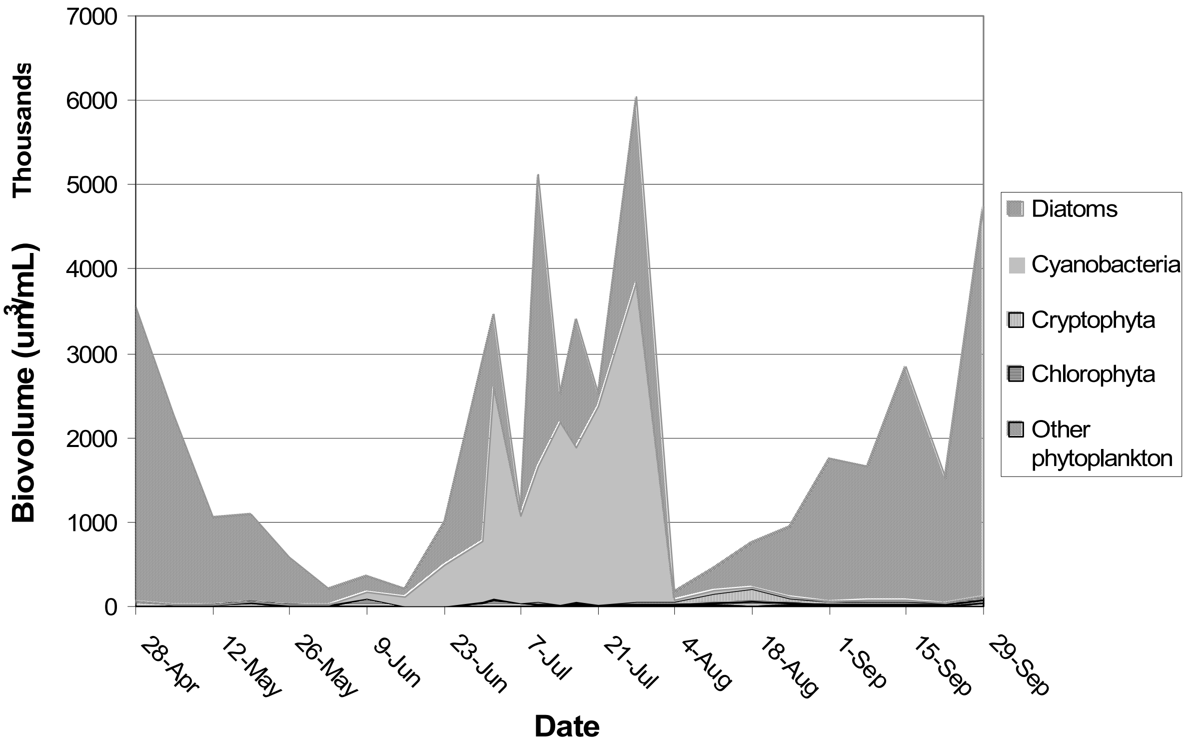

Figure 15a. Phytoplankton biovolume during 2003 (Diamond Lake surface - North End). Data source Aquatic Analysts 2003.

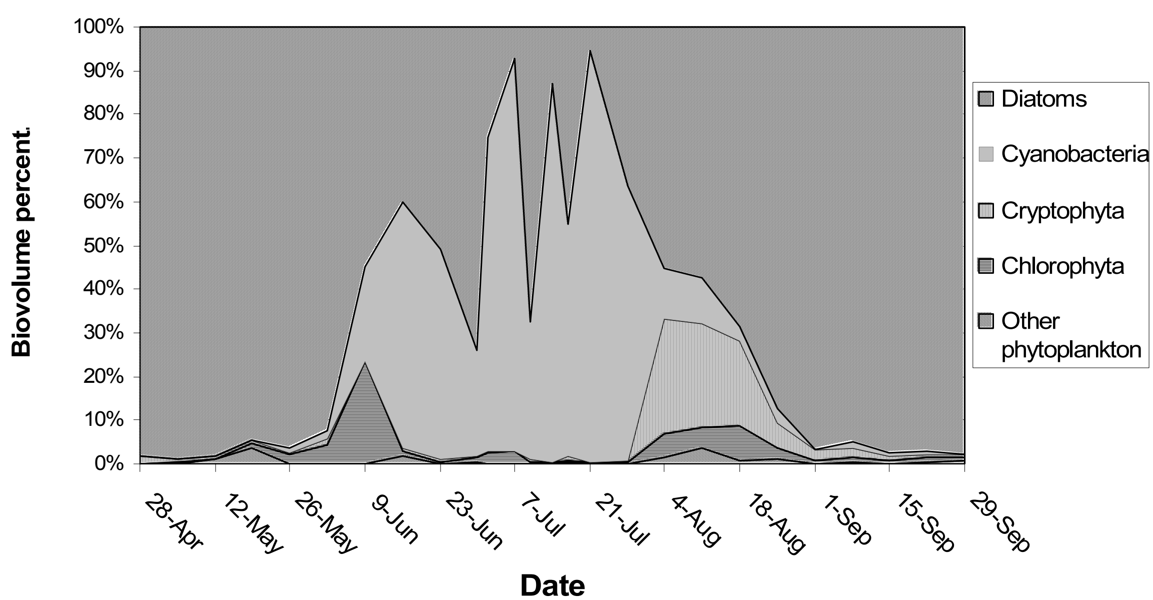

Figure 15b. Phytoplankton percent composition during 2003 (Diamond Lake surface - North End). Data source Aquatic Analysts 2003.

Lauer et al. (1979) reported *Anabaena circinalis* to be the dominant cyanobacterium. The earliest date that *Anabaena circinalis* dominated the phytoplankton community was on June 18, 1973. It was reported to have typically reached high numbers by July or August of each year during the study and frequently was the most numerous type of phytoplankton during September and October. Sanville and Powers (1971) reported 15,000 *Anabaena* cells/mL from a surface sample collected on September 27, 1971.

Based on analysis of sediment cores from Diamond Lake, Eilers et al. (2001a) reported that a shift in the planktonic diatom community of Diamond Lake occurred with the species *Fragilaria crotonensis* becoming much more abundant after approximately 1920 and remaining abundant through the 1950s, indicating more productive conditions in the lake. Although Eilers et al. (2001a) were not able to precisely date changes in the phytoplankton composition of the lake, these investigators found the diatom *Asterionella formosa* became somewhat more abundant near the year 1980 and that increase in the abundance of both *F. crotonensis* and *A. formosa* indicates a shift to more productive conditions in the lake. Sediment core analysis also indicated a 15-fold increase in the planktonic cyanobacteria population during the 20^{th} century (Eilers et al. 2001b). One genera of filamentous cyanobacteria, *Gloeotrichea*, was reported to have shown two major population increases first appearing in large numbers around 1920, following the introduction of trout in 1910 and another large increase near the time of the first rotenone treatment of Diamond Lake in 1954. The cyanobacteria species *Anabaena flos-aquae* and *Anabaena circinalis* were found at low densities prior to the introduction of fish in Diamond Lake. However, their abundance increased greatly around the year 1954 followed by a decrease to relatively low levels during the 1960s and 1970s with another large increase observed beginning in the 1990s. Peak concentrations of *Anabaena flos-aquae* found in samples during the summers of 2001 and 2002 were estimated to be approximately 1 million cells/mL. During the summer of 2003, the peak reported concentration of *A. flos-aquae* was 255,567 cells/mL on July 28. The high concentrations of *Anabaena* that have been recorded under bloom conditions in recent years correspond to periods of high tui chub abundance. In addition, during the summer of 2003, Jim Sweet (Aquatic Analysts, unpublished data) noted the rare occurrence of the cyanobacterium *Microcystis aeruginosa* in six samples however at much lower concentrations than *Anabaena*.

Evidence from the remains of the phytoplankton assemblage from the sediments was found to be consistent with other monitoring data indicating a decline in the water quality of Diamond Lake particularly over the last several decades (Eilers et al. 2001b). This decline is consistent with the idea that fish stocking and high populations of tui chub have contributed to this changed condition. Multiple mathematical models were used by Eilers et al. (2003) to assess current, past, and possible future ecological and water quality conditions in Diamond Lake. The models showed that the current high biomass of tui chub in the lake can largely explain the frequency of cyanobacterial blooms. The models also indicated that the intensity of the blooms can be strongly influenced by weather conditions. Extended periods of high temperatures, low wind speeds, and abundant solar radiation favor the development of a cyanobacterial bloom. Modeling by Eilers et al. indicated that to meet water quality goals in Diamond Lake it would be necessary to remove a high percentage (~90 to 100 percent) of the tui chub. These investigators estimated that the removal of tui chub from the lake would result in a reduction in the average peak biomass of cyanobacteria (averaged over 8 years) from approximately 20,000 kg to 4,000 kg. Modeling scenarios indicated that if tui chub were entirely removed from the lake, the frequency and intensity of cyanobacteria blooms would likely decline by approximately 80 percent, however under favorable weather conditions, some modest blooms would still be expected to occur.

Cyanobacteria frequently dominate the phytoplankton of productive lakes during the summer season. Environmental factors that contribute to the dominance of blue-green include: (1) a stable water column; (2) warm water temperatures; (3) high nutrient concentrations near the surface (particularly phosphorus); (4) high pH; (5) relatively low concentrations of carbon dioxide (CO_2); and, (6) low grazing pressure from large zooplankton (Zurawell 2000-01). Many filamentous planktonic cyanobacteria (including *Anabaena*) withstand periods of adverse environmental conditions by the production of specialized cells called akinetes that persist as a resting stage in the surface sediments of lakes until conditions are favorable for growth. Studies have shown that the rapid increase in the planktonic cyanobacteria population of a lake can result, in part, from recruitment of cyanobacteria from the sediments (Head et al. 1999).

All phytoplankton require nitrogen in relatively high amounts for optimal growth. Some cyanobacteria however have an advantage over other kinds of phytoplankton under conditions of high phosphorus and low nitrogen. Before nitrogen can be used in the synthesis of biological molecules it must be in the "fixed" (combined) form of ammonia or nitrate. Unlike other kinds of phytoplankton, many species of cyanobacteria, including *Anabeana*, have the ability to fix nitrogen gas (N_2) dissolved in the water. When the total nitrogen to total phosphorus ratio falls below approximately 14 to 1 by mass, the low availability of nitrogen generally favors the growth of nitrogen fixing cyanobacteria over other kinds of phytoplankton (Smith and Bennett 1999).

During the summer when phytoplankton abundance is high, densities are typically greatest near the surface where sufficient sunlight for photosynthesis is available and water temperatures are warm. Some species of cyanobacteria (including *Anabaena*) can gain a competitive advantage over other types of phytoplankton by their ability to regulate buoyancy through the production of intracellular gas vesicles. These gas vesicles allow the cells or colonies to migrate vertically through the water column and occupy a position with optimal light and nutrient concentrations. If calm wind conditions develop over a short period of time however, excess gas vesicles can cause the cells or colonies to rise to the surface where dense surface accumulations can develop. Once on the surface, exposure to high intensity light and possibly depletion of inorganic carbon can inhibit photosynthesis in the cyanobacteria cells and interfere with their ability to regulate their buoyancy. Winds can blow surface accumulations toward shore where dense surface scums can develop. Due to their buoyant nature, cyanobacteria concentrations near the water surface can change rapidly over a brief period of time. The rapid death and decay of cyanobacteria blooms under conditions of high surface concentrations can lead to the release of ammonia and depletion of dissolved oxygen in the water. Although these factors can be severe enough to result in fish kills, there are no known reports of this occurring in Diamond Lake.

Blooms of cyanobacteria are frequently associated with a development of undesirable conditions. In addition to imparting an unpleasant taste and odor to water, many kinds of cyanobacteria are known to produce potent nerve or liver toxins and blooms have caused illness and death in wildlife and livestock in many regions of the world. In addition, cyanobacteria toxins have been found to cause adverse health effects for humans (Falconer 1996). Studies have also shown that toxins from cyanobacteria have the potential for adverse effects on macrophytes, zooplankton and other aquatic species (Christoffersen 1996). These toxins have also been shown to suppress the growth of other types of algae possibly giving the cyanobacteria a competitive advantage.

The cyanobacteria species *Anabaena flos-aquae*, *Anabaena circinalis* and *Microcystis aeruginosa* are known to be potentially toxin-producing. Analysis results from samples taken from Diamond Lake during periods of high *Anabaena flos-aquae* abundance have been found to contain the neurotoxin anatoxin-a. Human health guidance from Yoo et al. (1995) and Chorus and Bartram (1999) indicate that lake users should avoid water contact at cyanobacteria densities above 15,000 cells/mL. The buoyant characteristic of *Anabaena* cells can lead to the formation of high densities on the lake surface or along shorelines where during bloom conditions concentrations can increase greater than 1,000 fold (Chorus and Bartram 1999). As mentioned previously, samples taken in Diamond Lake during blooms of *Anabaena* exceeded this concentration at times during the summer seasons during the 1970s. However during the recent summers of 2001, 2002, and 2003, *Anabaena* cell densities have been dramatically higher, greatly exceeding the threshold of 15,000 cells/mL. Due to the risk to public health, the Umpqua National Forest in cooperation with the Oregon Health Division and Douglas County Health Department restricted water contact activities in Diamond Lake during periods of high *Anabaena* abundance in each of these summers.

Studies have shown that the presence of fish that feed on zooplankton have a major influence on phytoplankton biomass and phytoplankton community structure (Lynch and Shapiro 1981). Shapiro et al. (1975) introduced the concept of "biomanipulation" as a management tool referring to manipulations of predator/prey relationships at the top trophic levels to influence aspects of a lake's productivity including reducing undesirable algae blooms. Biomanipulation is based on the idea that when the number of fish feeding on herbivore zooplankton are reduced, the abundance of large zooplankton species increases, resulting in an increase in the effectiveness of grazing on phytoplankton. This can reduce the density of phytoplankton and lead to improved water quality. Also, manipulations resulting in an increase in the number of fish that prey on fish that consume zooplankton can decrease predation on large zooplankton and result in an increase in the effectiveness of grazing on phytoplankton. Carpenter et al. (1985) used the term "trophic cascade" to describe trophic level interactions including fish or invertebrate predation that can lead to changes in the structure of zooplankton communities and alter the effectiveness of zooplankton grazing on phytoplankton. Based on numerous investigations, Wetzel (2001) concluded that the concept of cascading trophic interactions frequently fails in natural systems due to multiple compensatory mechanisms that occur rapidly after predator alterations. When aquatic ecosystems are altered, productivity is displaced and is not largely reduced or lost. However, an important management consideration is that the shifts in productivity can potentially be manipulated to a more desirable type viewed as beneficial for human uses of water (Wetzel 2001).

Drenner and Hambright (1999) reviewed the results from 41 biomanipulation experiments from 39 different lakes to determine if biomanipulation succeeded in improving water quality parameters including increased clarity, reduced phytoplankton biomass, and lower cyanobacteria density. These investigators reported that predacious fish stocking as a biomanipulation approach had the lowest success and partial fish removal was the most successful. Although they found differences in the results depending on different strategies, overall they found 61 percent of these experiments were consistently successful and that water quality was most likely to be improved and maintained where manipulations increased the abundance of *Daphnia* (a type of large bodied filter feeding zooplankton) and macrophytes. Drenner and Hambright (1999) concluded each type of biomanipulation they reviewed had been used successfully to improve water quality. However, due to the small number of biomanipulation techniques evaluated and the variation in the fish and plankton communities of different lakes, the best biomanipulation approach for a particular lake could not be identified.

Meronek et al. (1996) reviewed the results of 250 fish control projects contained in 131 papers. These researchers concluded that the total elimination of the targeted fish species was more successful than partial reduction of the targeted species in the majority of projects. Combined chemical and physical methods were reported to be successful in 66 percent of the projects evaluated. Stocking after combined chemical and physical control methods may have resulted in additional benefits to improve the rate of success for some projects. Meronek et al. found that the success rate for projects that used only physical control methods (e.g. nets, traps, electro-fishing, or a combination of physical treatments) ranged from 33 to 57 percent.

Although smaller than Diamond Lake, Lava Lake, a 332 acre lake located in the Cascade Mountain Range of Oregon has many characteristics similar to Diamond Lake. Since the 1970's, there has been and active mechanical harvesting program at the lake to reduce the tui chub population. In recent years it has been reported that approximately 10,000 pounds of tui chub are netted from the lake annually. Despite the effort made to reduce the tui chub population through mechanical removal these efforts have failed to significantly reduce the population. Lava Lake continues to experience blooms of cyanobacteria (*Anabaena*) and violations of state water quality standards. It has been suggested that the mechanical harvest of tui chub in Lava Lake maintains the population in an active growth mode and stimulates the recycling of nutrients favoring high phytoplankton density including blooms of blue-green algae (Eilers pers. comm.).

Studies have shown the important role of zooplankton in the regulation of phytoplankton biomass while at the same time encouraging its growth through phosphorus recycling (Villar-Argaiz et al. 2001). Sanni and Wærvågen (1990) reported significant increases in water quality and reduced summer cyanobacteria abundance following a rotenone treatment of a eutrophic lake in Norway. Lower nutrient concentrations and increased transparency were reported the first summer after the rotenone treatment and preliminary results from the second summer indicated further improvements. These investigators reported a 30 percent decrease in total phosphate concentrations the first summer after treatment compared to the mean of the preceding eight years without reductions in external loading. Sanni and Wærvågen concluded important factors in the decline of cyanobacteria abundance in the summer was not only increased grazing by *Daphnia* but also an increase in the rate of phosphate cycling benefiting phytoplankton species better able to utilize the nutrient supply from zooplankton excretion. Similarly, Prejs et al. (1997) reported that following rotenone treatment of a eutrophic lake in Poland, significant increases in water quality were observed. These investigators found that in the years following treatment, water transparency was 30 percent higher and there was a 2.8 fold decrease in algal biomass. The increase in water quality was largely attributed to an increase in the density of *Daphnia*.

Based on differing experiences with the application of biomanipulation and results of numerous studies, Reynolds (1994) reviewed the theoretical aspects of trophic level interactions, population responses, and shift of resources associated with these projects. Reynolds concluded that the reduction of algal biomass through biological methods depends upon an efficient and responsive filter feeder. In general, successful biomanipulation projects were dependent on the ability of *Daphnia* to respond to increases in available resources. Reynolds also reported that a dense population of benthic and peripheral macrophytes is an important factor associated with the successful application of biomanipulation. A dense macrophyte population provides a refuge from predators and is a source of bacterial decomposition of organic material which can provide the basis for an alternative source of nutrition to sustain at least a maintenance level of filter feeders.

Despite a large number of studies investigating the suitability of filamentous cyanobacteria (e.g. *Anabaena*) as a food source for zooplankton herbivores, the results of these studies remain largely inconclusive. Different studies have produced contradictory results even when the same cyanobacteria species were considered (Gliwicz 1990). Even though cyanobacteria have often been considered an unsuitable food for zooplankton grazers, in productive lakes where fish that fed on zooplankton (planktivores) died or where removed, it has been observed in many cases that the absence of fish results in an increase in the size and number of filter feeding zooplankton and this increase is associated with a decline in cyanobacteria density (de Bernardi and Giussani 1990).

Studies have shown that as long as cyanobacterial densities are not too high, grazing by zooplankton can keep cyanobacterial densities at reduced levels (Dawidowicz 1990, Christoffersen et al. 1993, Declerck et al. 1997). Based on a number of whole lake observations and enclosure experiments in a lake in southern California, Sarnelle (1993) concluded that the effects of herbivorous zooplankton on algal succession cannot be predicted from the relative susceptibilities of these algal species to grazing. Intense grazing by *Daphnia* during the spring-bloom period was observed to retard further succession to grazing resistant, filamentous cyanobacteria.

The cyanobacteria-zooplankton interactions in lakes are likely the result of a combination of factors including: the concentration, edibility, toxicity, and nutritional value of the cyanobacteria along with the degree the cyanobacteria mechanically interfere with filtering and the availability and nutritional value of other food sources (Gilbert and Durand 1990). In their review of numerous studies concerning the suitability of cyanobacteria as a food source for zooplankton, de Bernardi and Giussani (1990) concluded that even when cyanobacteria alone are not an adequate food source for zooplankton they can be an important complementary source of nutrition and when combined with other environmental factors zooplankton grazing can affect cyanobacteria density. Based on a number of short-term grazing trials, Epp (1996) concluded that zooplankton grazing by the large bodied cladoceran *Daphnia* promoted or maintained cyanobacteria dominance while decreasing the absolute abundance of cyanobacteria and phytoplankton as a whole. Based on these results and a review of other studies, Epp concluded that filamentous cyanobacteria should not be assumed to be resistant to zooplankton grazing until it has been evaluated for a particular lake. In Diamond Lake large populations of *Anabaena* over the past decade have coincided with a period of high tui chub abundance suggesting that increased predation of large zooplankton by tui chub has contributed to the severity of cyanobacteria blooms in Diamond Lake.

PRIMARY PRODUCTION

Primary production is the rate new organic matter is formed primarily through photosynthesis. Primary production occurs in the water column by phytoplankton and on the bottom of the lake by attached plants and algae. Estimates of phytoplankton primary production have been obtained for Diamond Lake by measuring the rate of inorganic carbon assimilation using a carbon-14 radioisotope. No known studies have investigated primary production of macrophytes or other attached photosynthetic organisms.

Phytoplankton primary production in Diamond Lake is typically highest during July and August and corresponds with the peak in phytoplankton abundance as measured by the concentration of chlorophyll *a*. Peak productivity during this time period occurs near the surface at approximately 2 m depth. Phytoplankton primary production is lower at other times of the year and is more uniformly distributed through the water column. Figure 16 displays typical profile data showing the variation in phytoplankton primary production by time and depth in Diamond Lake.

Although many studies have focused on the availability of nutrients as a regulator of productivity in lakes, the concept of cascading trophic interactions as proposed by Carpenter et al. (1985) has been suggested to account for the differences in primary productivity between lakes with similar nutrient availability but different food webs. Management activities that alter the food web structure may result in changes to the abundance or type of phytoplankton however this change is primarily a temporary shift of nutrients to other components of the ecosystem (Wetzel 2001).

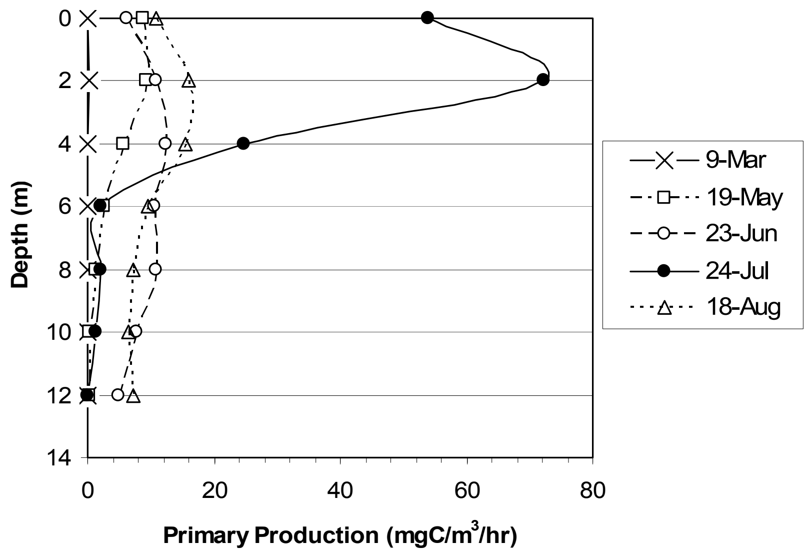

Figure 16. Phytoplankton primary production variation over time and depth during year 2002 (data from Salinas 2002).

ENVIRONMENTAL EFFECTS ON PHYTOPLANKTON AND PRIMARY PRODUCTION

Direct Effects:

Alternatives 1 and 4 would have no direct effects on the phytoplankton of Diamond Lake. Direct effects from a rotenone treatment or other activities proposed under Alternatives 2, 3, and 5 would likely have small to negligible effects on the phytoplankton of the lake. In a review of numerous studies of the direct effects of rotenone on phytoplankton, Bradbury (1986) found that most investigations showed that phytoplankton is not directly affected by rotenone at concentrations of up to 3 parts per million (ppm) of the 5 percent powdered form. Exceptions were one study where the chrysophyte, *Dinobyron,* was absent for a period of two weeks following rotenone treatment in a Montana pond treated with 0.7 ppm Pro-Noxfish®. Additionally, concentrations of 5 percent rotenone above 2 ppm killed all of the Chlorophyte *Volvox* and 1 ppm killed the dinoflagellate *Ceratium*. However, none of these algae types comprise a large portion of the phytoplankton community of Diamond Lake.

Under Alternative 2, 3, and 5, the connected actions proposed by Diamond Lake Resort (described in Chapter 2) would occur only after the affected areas are above the water level of the lake and would have no direct effects on the phytoplankton or primary productivity of the lake.

Indirect Effects:

Under Alternative 1 no change in the abundance or species composition of the phytoplankton assemblage or primary production would occur. Summer phytoplankton primary production would continue to be high and severe summer blooms of potentially toxic cyanobacteria would likely continue. During the summer blooms of toxin producing blue-green algae, adverse effects could occur to wildlife and domestic animals and could also result in adverse effects to other aquatic organisms.

Alternatives 2, 3, and 5 would have indirect effects on phytoplankton potentially a short period of time after mechanical fish removal begins. As the tui chub population is reduced by mechanical removal, it is likely the zooplankton population would increase, including large bodied species. The extent to which predation on zooplankton would be reduced depends on the degree that mechanical removal was successful at reducing the tui chub population. If mechanical removal successfully removes a significant portion of the fish population during the spring and early summer, the relatively large bodied zooplankton species could rapidly increase in number potentially reducing the phytoplankton density. High rates of zooplankton grazing could reduce the phytoplankton biomass to levels below average values observed over the last decade with a corresponding increase in water clarity and reduced epilimnetic pH values.

Bradbury (1986), in a review of the effects of rotenone treatment in lakes, found that the greatest reduction in zooplankton density occurred within 15 minutes and after one hour from the time of treatment and that within a few days, 95 to 100 percent of the open-water zooplankton are eliminated. Under Alternatives 2, 3, and 5, similar results would be expected in Diamond Lake after rotenone application with a severe reduction or elimination of the zooplankton population a short time after treatment. As is common following rotenone application, an algae bloom would be expected to occur and the bloom would most likely be dominated by cyanobacteria and/or diatoms. The cause of the bloom would be a result of reduced grazing on phytoplankton and the release of nutrients from fish carcasses and possibly sediments. Studies of other lakes have shown that algae levels typically increase 4 to 6 fold shortly after rotenone treatment compared to levels in years without treatment (Bradbury 1986). The bloom would most likely last for one or two months before subsiding as would be expected to occur with the onset of winter. The increase in phytoplankton biomass would result in decreased water quality, including a decrease in clarity and elevated pH values during the fall algae bloom.

Generally, zooplankton will return to a lake in large number within 2 to 10 weeks following rotenone treatment (Bradbury 1986). Under alternatives 2, 3, and 5 in the spring following the rotenone treatment, it is likely that a large diatom bloom would occur as would be expected to occur in a normal year. If the recovery of the zooplankton population is delayed, phytoplankton density would likely be greater than normal the following spring. It is likely the zooplankton population would rapidly increase in the spring following the spring diatom bloom as a response to the abundant food resources and no predation pressure from fish. Since large bodied zooplankton species are most susceptible to predation by fish, the relative number of large bodied zooplankton would be favored under the fishless condition of the lake. The high numbers of zooplankton could significantly graze down the phytoplankton population resulting in a significant reduction in phytoplankton biomass during the summer season. Under Alternatives 2, 3, and 5, it is likely that grazing by large-bodied zooplankton such as Dapnia would lead to a reduction in the density of cyanobacteria during the mid-summer season compared to other years. This effect would be expected due to either zooplankton grazing on the cyanobacteria or altering environmental conditions such as nutrient ratios or light availability creating favorable conditions for the growth of other types of phytoplankton. Under Alternatives 2, 3, and 5, it is possible recovery of the zooplankton population could be delayed resulting in water quality improvements occurring over a 3 year period.

Differences in fish stocking strategies implemented under Alternatives 2, 3, and 5 could result in different indirect effects. If the number of fish stocked under Alternative 2 resulted in heavy zooplankton losses due to predation an increase in phytoplankton density and possibly increased summer cyanobacteria blooms would be more likely to occur. Under Alternative 3, zooplankton populations would be less likely to be severely impacted by the stocking of large numbers of the type of domesticated trout proposed for stocking because these fish are not likely to prey significantly on zooplankton and therefore predation pressure on large bodied zooplankton would remain relatively low. Similar to Alternative 2, the rainbow fingerlings stocked under Alternative 5 have the potential to reduce the zooplankton population by predation. Under any action alternative however, salmonid stocking would be monitored to ensure that the number of fish stocked would not result in severe predation on the zooplankton population. Therefore, fish stocking under Alternatives 2 and 5 would not be likely to result in a decline in water quality however, Alternative 3 may have a slightly lower risk of adverse effects of fish stocking leading to a decline in water quality.

For Alternatives 2, 3, and 5, if tui chub remain following the rotenone treatment or if/when they are reintroduced into the lake in the future and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, it is assumed that the likelihood of sustaining improvements in the water quality over time may be increased with annual implementation of the described contingency plan.

As previously mentioned, under Alternative 2, 3, and 5, the operators of the Diamond Lake Resort have proposed activities within the draw down zone of the lake while this area is above the water level of the lake. Since the area proposed for these actions is relatively small and no in-water activities would occur, no indirect effects on phytoplankton or primary production from these activities are anticipated.

Because Alternative 4 would be implemented over a 7-year period, the effects on phytoplankton would occur over an extended time period. Following the initiation of mechanical fish removal, predation pressure on zooplankton would be reduced resulting in increased grazing on phytoplankton by zooplankton over the summer. Similar to Alternatives 2, 3, and 5 the degree to which the zooplankton would be able to reduce phytoplankton densities would depend on the extent mechanical fish removal significantly lowers predation on zooplankton by fish. No toxicants would be used under this alternative and as a result the zooplankton population would not be directly reduced by any treatment measures during implementation. Consequently, grazing on phytoplankton would be maintained and expected to increase as the tui chub population is reduced. Since the tui chub population would be reduced, but not eliminated from the lake under Alternative 4, and because the recent Lava Lakes data (Eilers pers. comm.) brings additional uncertainty regarding long-term water quality outcomes, there is a higher risk that this alternative would be less effective at sustaining water quality improvements into the future, and thus, reduced clarity, high pH, and increased abundance of cyanobacteria may recur. However, it is assumed that the likelihood of achieving or maintaining improvements in the water quality in the long-term may be increased with annual implementation of the described contingency plan.

Under Alternative 1, primary production by macrophytes and phytoplankton would remain unchanged. Alternatives 2, 3, and 5 would lower the lake level beginning with the fall/winter draw down period and the water level would remain low through the following summer, the lowered level along with increased phytoplankton grazing (resulting in a reduction in phytoplankton abundance as an indirect effect of mechanical fish removal) would potentially allow light to be transmitted to portions of the lake bottom that typically would be beyond the range where sufficient light would be available for photosynthesis. Although this could increase the area receiving sufficient light for photosynthesis by macrophytes and attached algae resulting in a shift of primary production from phytoplankton to bottom dwelling plants, total primary production in the lake would be offset by the potential loss in primary production from the macrophytes in the areas de-watered during the summer draw down. Under Alternatives 2, 3, and 5, phytoplankton density and primary production would be expected to be lower approximately three years following the rotenone treatment. Implementation of Alternative 4 would likely result in a slight gain in primary production of macrophytes after a period of approximately 7-years. There would be the possibility that this increase could not be maintained over time because under Alternative 4, there would be a higher probability that the tui chub population would increase significantly over time indirectly leading to an increase in phytoplankton abundance. Additionally, under Alternative 4 there is a possibility that during the period of mechanical fish removal the tui chub population could remain in an active growth mode capable of significantly reducing the density of large zooplankton species resulting in relatively high phytoplankton density compared to Alternatives 2, 3, and 5. In the long-term under any action alternative, an increase in water clarity would result in a shift toward higher total primary production in macrophytes and attached algae and reduced levels in phytoplankton.

Cumulative Effects:

The composition of the phytoplankton population in Diamond Lake to some extent results from the cumulative effects of activities in the watershed that have the potential to be a source of nutrients to the lake. Although past developments within the watershed have likely contributed to nutrient enrichment of the lake to some extent, the majority of the nutrients in the lake originate from natural sources. Ongoing and past projects including fish stocking, installation and operation of the waste water diversion system and erosion control activities, have cumulatively impacted lake conditions and influenced the abundance and species composition of the phytoplankton population. Under Alternative 1, effects from activities within the watershed (e.g. summer home use and erosion various projects) could influence the phytoplankton community composition of the lake resulting in negative cumulative effects on water quality. Any of the action alternatives in combination with ongoing, past projects (e.g. wastewater diversion system), and reasonably foreseeable actions have the potential to cumulatively lead to changes in the phytoplankton composition and primary productivity with positive influences on lake water quality. See EIS Tables 9-11 for a complete list of projects.

Conclusions:

Under Alternative 1, the tui chub population would remain high, resulting in fewer filter feeding zooplankton and a corresponding high rate of phytoplankton production. Summer eplimnetic pH values would remain high and likely above state water quality standards. A high biomass of phytoplankton during summer algae blooms would continue to reduce water clarity and toxin producing blue-green algae blooms would be expected to be more frequent and severe than would occur under any of the action alternatives.

All action alternatives have the potential to affect the abundance and species composition of the phytoplankton population. Implementation of either Alternative 2, 3, or 5 would result in changes to the phytoplankton assemblage in the shortest time period due to the elimination of fish through the rotenone treatment followed by an increase in the zooplankton population. Water quality would be expected to improve within a 3 year time period under these alternatives. The long-term effects on the phytoplankton under Alternatives 2, 3, and 5 would depend on the fish stocking strategy adopted. Monitoring of lake conditions under any of the action alternatives would ensure fish stocking levels that do not result in severe predation pressure on zooplankton. The fish stocking strategy proposed under Alternative 3 would likely have small effects on the zooplankton population and therefore may have a lower risk that predation on zooplankton would slow water quality recovery.

Under Alternative 4, changes to the phytoplankton assemblage would likely occur over time as the tui chub population is reduced and predation on zooplankton is reduced. Water quality would be expected to improve after the 7-year period of mechanical tui chub removal. The long-term effects on the phytoplankton under Alternative 4 would depend on the continued success of mechanical removal techniques at limiting the tui chub population and the effectiveness of annual tui chub removal indirectly reducing the severity of cyanobacteria blooms over time. If mechanical harvest methods employed under Alternative 4 failed to maintain a significantly reduce the tui chub population, it is possible the tui chub population would remain in an active growing mode that would continue to impact the zooplankton assemblage of the lake resulting in relatively high phytoplankton densities and poor water quality compared to other action alternatives.

Implementation of any of the action alternatives would have short-term impacts at the 6th field scale and would meet Aquatic Conservation Strategy Objectives in the long-term at local and broader scales. Action alternatives specifically contribute positively toward meeting Objective 4[19], to maintain and restore water quality. Improvements in water quality would include lower phytoplankton biomass, lower epilimnetic pH values, and increased clarity.

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[19] ACS Objective 4 – Maintain and restore water quality necessary to support healthy riparian, aquatic, and wetland ecosystems. Water quality must remain within the range that maintains the biological, physical, and chemical integrity of the system and benefits survival, growth, reproduction, and migration of individuals composing aquatic riparian communities.

Table 5 provides a summary of important conclusions from this section and a comparison of the alternatives effects on water quality.

Table 5. Summary of Alternative measures on phytoplankton density and primary production, and potential cyanobacteria toxin production.

Alternatives	Alternative 1 – No Action	Alternative 2 -Rotenone Put, Grow and Take Fishery	Alternative 3 – Rotenone Put and Take Fishery	Alternative 4 - Mechanical & Biological	Alternative 5- Modified Rotenone and Fish Stocking
Measure **Phytoplankton Density and Primary Production**	**Short-term** Phytoplankton density would remain high and continue to degrade water quality.	**Short-term** During the first 3 years after treatment, phytoplankton density and primary production potentially would remain high and result in poor water quality.	**Short-term** During the first 3 years after treatment, phytoplankton density and primary production potentially would remain high and result in poor water quality.	**Short-term** For 7 years phytoplankton density would remain high. Near the end of 7 years of treatment, phytoplankton density expected to decrease and result in improved water quality.	**Short-term** During the first 3 years after treatment, phytoplankton density and primary production potentially would remain high and result in poor water quality.
(Qualitative Trend Analysis)	**Long-term** Phytoplankton density would remain high and continue to degrade water quality.	**Long-term** After 3 years following treatment, phytoplankton density and primary production expected to decrease and result in a noticeable improvement in water quality. At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.	**Long-term** After 3 years following treatment, phytoplankton density and primary production expected to decrease and result in a noticeable improvement in water quality. At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.	**Long-term** After 7 years of treatment, phytoplankton density and primary production expected to be lower for a period of time resulting in improved water quality over the existing condition. However, if annual mechanical removal fails or is stopped, the tui chub population would rebound and subsequent increases in pH and declines in water quality are expected. The likelihood of achieving or maintaining improvements in the water quality in the long-term would be increased with annual implementation of the described contingency plan over time.	**Long-term** After 3 years following treatment, phytoplankton density and primary production expected to decrease and result in a noticeable improvement in water quality. At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur. However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.

Table 5. Summary of Alternative measures on phytoplankton density and primary production, and potential cyanobacteria toxin production (continued).

Alternatives	Alternative 1 – No Action	Alternative 2 -Rotenone Put, Grow and Take Fishery	Alternative 3 – Rotenone Put and Take Fishery	Alternative 4 - Mechanical & Biological	Alternative 5- Modified Rotenone and Fish Stocking
Measure **Expected Blue-Green Algae Toxin Production**	**Short-term** Blue-green algae toxin production expected to continue. Annual lake closures expected.	**Short-term** During the first 3 years after treatment, blue-green algae toxin production potentially would remain high. Annual lake closures still expected.	**Short-term** During the first 3 years after treatment, blue-green algae toxin production potentially would remain high. Annual lake closures still expected.	**Short-term** For 7 years blue-green algae toxin production would remain high and annual lake closures are still expected. Near the end of 7 years of treatment, blue-green algae toxin production expected to decrease; severity of algae blooms would be reduced and annual lake closures not expected.	**Short-term** During the first 3 years after treatment, blue-green algae toxin production potentially would remain high. Annual lake closures still expected.
(Qualitative Trend Analysis)	**Long-term** Blue-green algae toxin production expected to continue. Annual lake closures expected.	**Long-term** After 3 years following treatment, blue-green algae toxin production expected to decrease. Although periodic lake closures are still possible, severity of algae blooms would be reduced and annual lake closures not expected. At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur and annual lake closures would again be expected However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.	**Long-term** After 3 years following treatment, blue-green algae toxin production expected to decrease. Although periodic lake closures are still possible, severity of algae blooms would be reduced and annual lake closures not expected. At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur and annual lake closures would again be expected However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.	**Long-term** After 7 years of treatment, blue-green algae toxin production expected to be lower for a period of time. However, if annual mechanical removal fails or is stopped, the tui chub population would rebound and annual lake closures would again be expected. The likelihood of achieving or maintaining improvements in the water quality in the long-term would be increased with annual implementation of the described contingency plan over time.	**Long-term** After 3 years following treatment, blue-green algae toxin production expected to decrease. Although periodic lake closures are still possible, severity of algae blooms would be reduced and annual lake closures not expected. At some unknown point in the future, if/when tui chub remain or are reintroduced and contingency plans fail, adverse impacts similar to current water quality problems would be expected to recur and annual lake closures would again be expected However, if/when tui chub recur the likelihood of sustaining improvements in the water quality over time would be increased with annual implementation of the described contingency plan.

/s/ Al Johnson